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Development of ecosystems under human activity in the North-East Estonian industrial region: forests on post-mining sites and bogs

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Development of ecosystems under human activity in the North-East Estonian industrial region: forests on post-mining sites and bogs

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CONTENTS

LIST OF ORIGINAL PUBLICATIONS

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I Pensa, M., **Karu, H**., Luud, A., Rull, E., Vaht, R. 2008. The effect of planted tree species on the development of herbaceous vegetation in a reclaimed opencast. *Canadian Journal of Forest Research* 38: 2674–2686.

II **Karu, H**., Szava-Kovats, R., Pensa, M., Kull, O. 2009. Carbon sequestration in a chronosequence of Scots pine stands in a reclaimed opencast oil shale mine. *Canadian Journal of Forest Research* 39: 1507–1517.

III Vaht, R., Pensa, M., Sepp, M., Luud, A., **Karu, H**., Elvisto, T. 2010. Assessment of vegetation performance on semicoke dumps of Kohtla-Järve oil shale industry, Estonia. *Estonian Journal of Ecology* 59: 3–18.

IV **Karu, H**., Pensa, M., Rõõm, E.-I., Portsmuth, A., Triisberg, T. 2014. Carbon fluxes in forested bog margins along a human impact gradient: could vegetation structure be used as an indicator of peat carbon emissions? *Wetlands Ecology and Management* 22: 399–417.

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	I	II	III	IV
Original idea		*		*
Study design		*		*
Data collection	*	*	*	*
Data analysis	*	*		*
Manuscript preparation	*	*	*	*

LIST OF ABBREVIATIONS

[K]	concentration of potassium (μg g^{-1})
[N]	concentration of nitrogen (%)
[P]	concentration of phosphorus (mg g^{-1})
ANOVA	analysis of variance
BA	stand basal area (m^2 ha^{-1})
C	carbon
CH_4	methane
CO_2	carbon dioxide
DBH	stem diameter at breast height (cm)
MANOVA	multivariate ANOVA
NMDS	nonmetric multidimensional scaling
PAHs	polycyclic aromatic hydrocarbons
pH	hydrogen ion concentration; a measure of the acidity or alkalinity of a solution
pMC	percent modern carbon
R_s	soil respiration rate (g C m^{-2} day^{-1})
SOC	soil organic carbon
SOM	soil organic matter
TOC	soil total organic carbon
V	stem volume; growing stock (m^3 ha^{-1})
WTD	water table depth (cm)

I. INTRODUCTION

Mining and processing of minerals are a fundamental part of the economic activities of mankind. As the pressure of mining activities on the environment is increasing due to growing population and technological development, more attention has been given to the assessment and mitigation of the environmental effects associated with mining. One of the most serious impacts of mining is the replacement of multifunctional natural and cultural landscapes with new artificial landforms, like mine spoil and waste piles. The functioning of ecosystems could also be affected by change in hydrological conditions in the vicinity of mines and industrial pollution related to mining or processing of minerals. Degradation of ecosystems decreases our natural capital, as natural and cultural landscapes provide vital goods services to humankind (Constanza et al. 1997; de Groot et al. 2002; Millennium Ecosystem Assessment 2003). Functions or services provided by ecosystems include (de Groot and Hein 2006): 1) provisioning of resources, 2) regulation of biogeochemical cycles and other regulation functions, 3) habitat functions, i.e. maintenance of biodiversity and evolutionary processes, and 4) cultural and amenity functions. Assessing the impact of mining activities on ecosystem functions is important for balanced decision-making between alternative land uses in areas containing mineral resources. In post-mining sites, restoration measures should seek to maximize goods and services provided by novel emerging ecosystems, thus increasing our natural capital.

I.1. Establishment of forest ecosystems on mine spoil

Strip mining of natural resources causes large-scale disturbances of existing vegetation, soil and topography, as overburden covering the ore has to be removed and displaced. In most countries, legislation now demands that areas damaged by mining should be restored back to beneficial use. Therefore, post-mining sites are used for various purposes, such as forestry, agriculture, recreation, tourism, as nature reserves, residential or industrial lands etc. However, ecological restoration attempts to return degraded ecosystems to their historic trajectory (SER 2004). The complete reconstruction of previous ecosystems from barren mine spoil is rare, because of unfavourable conditions for plant growth and high cost of restoration measures. Another aim of ecologically sound restoration could be the establishment of ecosystems with as many functional elements as possible, substantial productivity according to local conditions, strong biotic interactions and little need for maintenance (Walker and del Moral 2003).

Common practice for reclamation of exhausted opencast has been to level the mine spoil and plant or sow fast-growing woody plants that minimize erosion and enhance soil development. Monospecific tree plantations on mine spoil promote the development of forest ecosystems and may also give high timber production, thus influencing positively the economy of the region (Singh et al. 2002). In recent decades, however, ecological aspects have gained larger signi-

ficance in restoration, especially that of sustainability and biodiversity (Bradshaw 1998). The use of monocultures for restoration has been criticised for their low stability, low resource-use efficiency and low level of biodiversity (Lugo 1997). Therefore, restoration methods based on natural processes with minimum intervention have been advocated, especially if conditions for plant growth are not very extreme (e.g. Tischew and Kirmer 2007; Prach and Hobbs 2008; Hobbs and Cramer 2008; Walker et al. 2014). It has been shown that spontaneous succession gives satisfactory results in most degraded sites in Central Europe, including on mine spoil (Prach et al. 2007, 2011, 2013, 2014). Spontaneous succession leads to the development of more diverse plant communities with species composition closer to that of (semi)natural vegetation compared to technically reclaimed (levelled and afforested) sites (Hodačová and Prach 2003; Pensa et al. 2004b; Mudrák et al. 2010).

The establishment of species during spontaneous succession depends on chance, habitat conditions and interactions between new and existing species (Dobson et al. 1997). At initial stages of succession, synanthropic species (i.e. ruderals and weeds) usually dominate the disturbed sites (Prach et al. 2013, 2014), but further trajectory of unassisted ecosystem development in opencasts is determined by surrounding plant communities, primarily by the availability of diaspores, macroclimate and local site factors (Prach and Řehounková 2006; Řehounková and Prach 2007; Tischew and Kirmer 2007). Soils in post-mining sites may possess several adverse properties for plant growth which slow down successional processes (Bradshaw 1983, 1997; Martínez-Ruiz and Marrs 2009). However, nutrient-poor open habitats, like mine spoil at early successional stages, may have high ecological value of their own, as they provide habitats for poor competitors like orchids and other slow-growing species, including protected species (Kirmer et al. 2008; Tropek et al. 2010, 2012; Tischew et al. 2014). Thus, traditional reclamation measures accelerating succession could reduce the ecological potential of post-mining sites (Kirmer and Mahn 2001; Tischew and Kirmer 2007; Tropek et al. 2010, 2012). In addition, unreclaimed sites with variable unlevelled substrate and diverse tree layer also provide more habitat niches for different understory species, promoting the development of diverse vegetation (Pensa et al. 2004b; Tischew and Kirmer 2007). If trees are planted, they act as ecosystem engineers, creating conditions that favour some understory species and exclude others (Jones et al. 1997; Mitchell et al. 2007). As a result, tree species selected for afforestation may strongly influence the development of understory vegetation (Dzwonko and Loster 1997; Mudrák et al. 2010; Thomaes et al. 2014).

Besides biodiversity, carbon (C) sequestration has also become an important environmental issue in light of climate change. Therefore, maximizing C sequestration should be one goal of rehabilitation of mine spoil in order to offset some of the C emissions caused by mining and burning of fossil fuels. However, the carbon budget of ecosystems emerging on former mining areas is poorly understood (Shrestha and Lal 2006). So far, there are very few publications providing concurrent estimates of change in biomass and soil C stocks

following reclamation of degraded land (Shrestha and Lal 2010; Amichev et al. 2008; Frouz et al. 2013). At the same time, it is important to know which ecosystem compartments contribute more to C sequestration, as different C pools vary greatly in C turnover rates. Since mean residence time for soil organic carbon (SOC) is several times higher compared to that of vegetation in high latitude forests (Mahli et al. 1999), restoration practices that maximize C sequestration by soils should be preferred.

Studies carried out on former agricultural lands show that following afforestation, soils accumulate less C and at a slower rate than the aboveground biomass (Richter et al. 1999; Hooker and Compton 2003; Thuille and Schulze 2006; Ouimet et al. 2007; Vesterdal et al. 2007). Since the C content of mine spoil is usually very low, soils developing on mine spoil have a large potential for C sequestration (Akala and Lal 2000; Ussiri and Lal 2005; Amichev et al. 2008; Vindušková and Frouz 2013) until a new steady state between litter input and decomposition is reached. C accumulation in soils after afforestation can be greatly affected by the selection of tree species (Jandl et al. 2007; Vesterdal et al. 2013; Frouz et al. 2009, 2013). One of the main factors by which tree species influence soil development is litter quality, as more easily decomposable litter promotes soil bioturbation by earthworms, resulting in soil organic matter (SOM) stabilization in soil aggregates (Lavelle et al. 1997; Reich et al. 2005; Frouz et al. 2009, 2013). Therefore, deciduous stands with higher litter quality may enhance C sequestration into more protected mineral soil horizons, whereas forest floor is thicker under conifers (Jandl et al. 2007; Turk et al. 2008; Frouz et al. 2009; Vindušková and Frouz 2013). Dominant tree species also substantially affect the soil chemical properties and microbial decomposition processes in post-mining sites (Šnajdr et al. 2013). A deeper understanding of the factors determining vegetation development and carbon dynamics in reclaimed mine spoils is needed in order to choose the optimal measure considering the restoration goals.

1.2. Estonian oil shale industry

In Estonia, oil shale mining has affected about 1% of the terrestrial area. Estonia accounts for about 70% of the world's oil shale production, although it has only 1% of the world's and 17% of Europe's oil shale resources (EASAC 2007). The largest deposits are found in the United States, Brazil, Russia, China, Australia and Jordan; presently oil shale is extracted in Estonia, Russia, China, Brazil, Australia and Germany (Kattai 2003).

The Estonian oil shale (kukersite) basin was formed in the Middle Ordovician and is covered by Ordovician and Devonian carbonaceous rocks and Quaternary sediments. The Estonian oil shale deposit lies in the western part of the Baltic Oil Shale Basin, in North-East Estonia, with an area of approximately 3,000 km^2 (Fig. 1) (Kattai and Bauert 2007). Mining of oil shale in Estonia started in 1918 and today mined areas cover about 430 km^2. The total area with

proved and probable reserves is ~1,700 km^2, while the oil shale on 850 km^2 is not taken into account in the Environmental Register of Estonia (Kattai and Bauert 2007).

The mineable oil shale bed is located at a depth of 10–70 m due to the southward dipping of the Ordovician strata. The thickness of the overburden material determines the mining method and the quality of the mine spoil. Opencast mining is preferred in the northern part of the deposit, up to a depth of 30 m. This area was previously covered mainly by forests and peatlands (Vaus 1970; Toomik and Kaljuvee 1994). In the southern part, oil shale is extracted in underground mines. Mining peaked in 1980 and then decreased substantially until 1999, after which there has been a steady increase (Hermet 2014). The ratio of oil shale production from opencast and underground mines is currently about 50:50, but the importance of underground mining is increasing since more easily accessible stocks have already been exploited and remaining stocks lie deeper (Hermet 2014). Over 80% of the mined oil shale is consumed in power plants – more than 80% of Estonian electricity is generated from this fossil fuel (Hermet 2014). Oil shale is also used to produce shale oil, coke, bitumen, cement, and a wide variety of chemical products.

Oil shale fired power plants have been the main source of air pollution in Estonia, although the situation has improved significantly since 1990 (Link 2014). Waste from oil shale industry (mining, thermal processing and combustion) comprise about 80% of the total amount of waste produced in Estonia and almost all of the hazardous waste (Kuusik and Hermet 2012). Hazardous substances are formed during energy and oil production and about half of the total mass of the raw material becomes waste after combustion (Hermet 2014). Semicoke is produced during oil retorting and is considered the most hazardous substance, since semicoke and leachates from it contain polycyclic aromatic hydrocarbons (PAHs), oil products, sulphates, phenols, cresols and other harmful pollutants (Truu et al. 2003). Old semicoke dumps in Kohtla-Järve and Kiviõli were serious pollution sources to the environment due to leaching of organic substances into rivers and aquifers (Truu et al. 2002), emission of PAHs, volatile phenols and dust (Orupõld and Henrysson 1999; Kundel and Liblik 2000; Raave et al. 2004), and self-ignition, causing the emission of hydrogen sulphide, ammonia, and heavy metals into the air (Raave et al. 2004; Oja et al. 2007). Therefore, these semicoke deposits were closed and recultivated in 2010–2014.

1.3. Development of forest ecosystems in post-mining sites

Several new landscape units have developed in North-East Estonia as a result of mining: there are mine spoils in exhausted opencasts, depressions above underground mines, piles of limestone waste, semicoke deposits and ash plateaus and hills near the power plants. Afforestation has been considered the most appro-

priate way of restoring the productivity of mine spoil and waste piles, as they are poor in organic matter and nutrients, and the substrates are unfavourable for plant growth (Vaus 1970; Kaar 2002b, 2003, 2010; Lõhmus et al. 2007). Afforestation of opencast mine spoils in Estonia started in 1959 and up to now more than 140 km^2 of land has been recultivated. Reclamation is carried out in two stages: first, the spoil heaps are levelled and left to stabilize for a couple of years; second, tree seedlings are planted on levelled mine spoil. The spoil material is alkaline and highly skeletal, containing 40–75% limestone (Reintam 2004). The organic carbon content in the fine earth fraction varies between 1.4 and 2%, but may attain 4–5% (Reintam 2004). Mostly Scots pine (*Pinus sylvestris* L.) has been used for reclamation, and pine plantations covered 85% of the afforested land by 2008 (Kaar 2010). The wide use of *P. sylvestris* for afforestation is due to the successful establishment of pine stands under the harsh conditions of mine spoil (Kaar 2002a, 2010). Other more frequently planted tree species include silver birch (*Betula pendula* Roth), Norway spruce (*Picea abies* (L.) Karst.), larch species (*Larix* spp.) and black alder (*Alnus glutinosa* (L.) Gaertn.) (Ostonen et al. 2006; Kaar 2010). In total, more than 52 woody species have been used in reclamation, mostly in small experimental plots. Older oil shale opencasts that were closed before the afforestation practice started have recovered by natural succession. Also, in the Narva mine about 200 ha of mine spoil was experimentally left unplanted and can be used to estimate the success of spontaneous vegetation recovery in exhausted oil shale opencasts.

So far, the establishment of plantations has been driven mainly by the aim to increase the productivity of disturbed areas rather than by the need for ecological restoration (Luud and Pensa 2004). It has been found that productive forest stands with unique species composition, moder-type forest floor and Calcaric Regosols or Calcaric Arenosols develop on mine spoil within 20–30 years after afforestation (Reintam and Kaar 2002; Reintam et al. 2002; Reintam 2004; Laarmann et al. 2015). However, there is still a lack of knowledge of and information on suitable tree species and methods for afforestation (Kuznetsova et al. 2009). Furthermore, only a few studies have considered aspects like biodiversity and C sequestration in post-mining sites and the sustainability of different reclamation options (Luud and Pensa 2004; Pensa et al. 2004b; Karu 2005; Laarmann et al. 2015).

On semicoke hills, the revegetation started in the early 1970s. By the time of the closing of the landfills, the planted area at Kohtla-Järve had reached 60 hectares; in addition, ~20 ha had naturally grown vegetation. The most productive species were broadleaved trees, such as *A. glutinosa* and *B. pendula*, while conifers did not perform well on the semicoke dumps (Kaar 2003). The development of plant cover significantly reduced the erosion and environmental pollution originating from the dumps (Kaar 2003; Truu et al. 2003); nevertheless, during the closing of old landfills, the planted and spontaneously developed vegetation was razed, except for small areas with well-developed ecosystems.

1.4. Impact of oil shale industry on mire ecosystems

Underground oil shale mining has no severe impacts on the aboveground forest ecosystems in North-East Estonia, except for some subsided areas, where forests have been replaced by wetlands and water bodies (Rull et al. 2005). In areas were geological conditions do not cause flooding of the depressions, draining caused by subsidence cracks has even increased the productivity of trees (Norway spruce) above mines, although tilting of trees leads to a change in wood quality (Läänelaid et al. 2009). In the near future, however, mining is approaching the protected mire areas of Muraka (Selisoo and Ratva bogs) and Puhatu mire complexes. At the same time, little is known of the effect of underground mining on the mires.

Mire ecosystems are considered important long-term sinks and pools for atmospheric carbon (Turunen et al. 2002; Mitsch et al. 2013), since CO_2 is accumulated as peat due to anoxic conditions slowing down the decomposition of plant organic matter. The northeastern part of Estonia is one of the most paludified areas of the country, where mires cover about 40% of the total area. Therefore, besides oil shale, a significant amount of Estonian peat production comes from this region.

Mining activities affect peatlands in several ways. Besides the direct destructive impact on the vegetation and landscape, mires are also severely influenced by lowering of the water table and by high levels of alkaline fly ash coming from oil shale fueled power plants (Karofeld and Ilomets 2008). Per each ton of oil shale mined in Estonia, 10–23 cubic meters of water must be pumped out of the mine area (Kattai and Bauert 2007), which disturbs the water regime of the surrounding land. The impact of drainage on northern peatlands has been widely studied, since during the last century the hydrological cycle of peatlands in Nordic countries has been extensively altered by drainage for forestry, agriculture and peat mining (Vasander et al. 2003). Persistent lowering of the water table induces secondary succession of mires towards forest-like ecosystems (Cedro and Lamentowicz 2008) and potentially enhances the decomposition of the peat layer, accompanied by higher soil CO_2 efflux (Moore and Dalva 1993; Martikainen et al. 1995; Silvola et al. 1996). At the same time, CH_4 emissions decrease after drainage (Martikainen et al. 1995; Nykänen et al. 1998; Minkkinen et al. 2007; Salm et al. 2009). Annual peat CO_2 emissions (Silvola et al. 1996) and also net CO_2 and methane fluxes from peatlands (Couwenberg et al. 2011) have been found to correlate well with average water table depth (WTD). However, the response of C fluxes to drainage is not uniform across peatland types and depends largely on local site conditions and on the severity of human impact (Ojanen et al. 2013).

Unlike other peatlands, bogs receive nutrients solely from atmospheric deposition, and therefore they are very sensitive to any additional influx of nutrients due to air pollution. It has been found that alkaline fly ash emitted by oil shale fueled power plants has adverse effects on the productivity of *Sphagnum* mosses and results in a shift in species composition in favor of vascular plants

(Karofeld 1996; Paal et al. 2010). In the 1990s, the power plants were equipped with effective dust filters, but despite the decreased pollution load in North-East Estonia in recent decades, bogs close to pollution sources still have low *Sphagnum* cover and a low proportion of bog-specific vascular plants (Karofeld et al. 2007; Paal et al. 2010).

Increased deposition of air-borne nutrients and alkaline dust also accelerate the growth of bog trees (Karofeld 1994; Pensa et al. 2004a; Kaasik et al. 2008), which has led to forestation of previously open or sparsely wooded bog areas. A more abundant nutrient-rich substrate and an increase in soil pH resulting from alkaline air pollution also creates more favorable conditions for bacterial activity and may lead to increased CO_2 emissions (Ivarson 1977; Bååth et al. 1992; Bragazza et al. 2006). However, the impact of air pollution on C fluxes has not been studied as extensively as the effect of drainage, and there are no studies on possible interactions with other disturbances.

The condition of mires is usually assessed by means of vegetation inventories. Some studies (Bubier et al. 1995; Dias et al. 2010; Couwenberg et al. 2011; Urbanová et al. 2012) propose that plant species composition could also be a good indicator of net CO_2 fluxes and methane emissions from peatlands, as it integrates several abiotic and biotic factors that influence C fluxes. In forestry-drained sites, methane emissions have been found to be correlated with stand volume (Minkkinen et al. 2007). At the same time, there is some concern that the response of vegetation to the changing environment may be delayed under certain conditions (Couwenberg et al. 2011). Empirical relationships between vegetation structure and soil C fluxes are still lacking, as carbon balance measurements are laborious. Therefore, reliable proxies for C emissions would significantly facilitate the assessment of the effect of disturbances on the peatland functioning.

1.5. Aims of the thesis

The first aim of this thesis was to describe the development of vegetation and soils in exhausted opencast oil shale mines and on semicoke hills, and how this process is influenced by the species selection used for reclamation. The consequences of afforestation are also compared to the natural succession in terms of biodiversity and productivity. The second aim was to assess changes in bog ecosystems, which are abundant in North-East Estonia, caused by alkaline air pollution from power plants and the lowering of the water table by human activities. Future research needs are identified for more precise estimation of the recovery of ecosystem services in post-mining sites and their deterioration in peatlands due to oil shale industry.

The key hypotheses set up and tested in the thesis:

1. Afforestation promotes tree productivity on oil shale mining spoil. Most of the C sequestrated by forest plantations established on mine spoil is accumulated in trees, but soils also contribute to a significant extent.
2. In reclaimed opencasts, growing stock, soil fertility and topsoil depth are higher under 30-year-old broadleaved (silver birch and alder species) stands compared to Scots pine plantations. Areas left to spontaneous succession develop significantly slower compared to afforested sites.
3. Tree species used for reclamation influence the development of understory vegetation on mine spoil and semicoke dumps.
4. Human activities in the North-East Estonian industrial area (drainage, mining and air pollution) cause changes in vegetation and C cycle in bog margins and initiate their succession toward forest-like ecosystems. Peat respiration rates are higher and CH_4 emissions lower in disturbed bogs compared to sites without direct human influence.

2. MATERIALS AND METHODS

2.1. Study sites

The study sites were located mainly in North-East Estonia (Fig. 1). Estonia is situated in the northwestern part of the East-European platform, in the hemi-boreal subzone of the boreal vegetation zone. Climatically, Estonia belongs to the temperate zone, in a transitional zone between areas experiencing maritime and continental climate. The long-term average monthly temperatures in north-eastern Estonia range from –6.1°C in February to 16.4°C in July; the long-term mean for total annual precipitation is 696 mm.

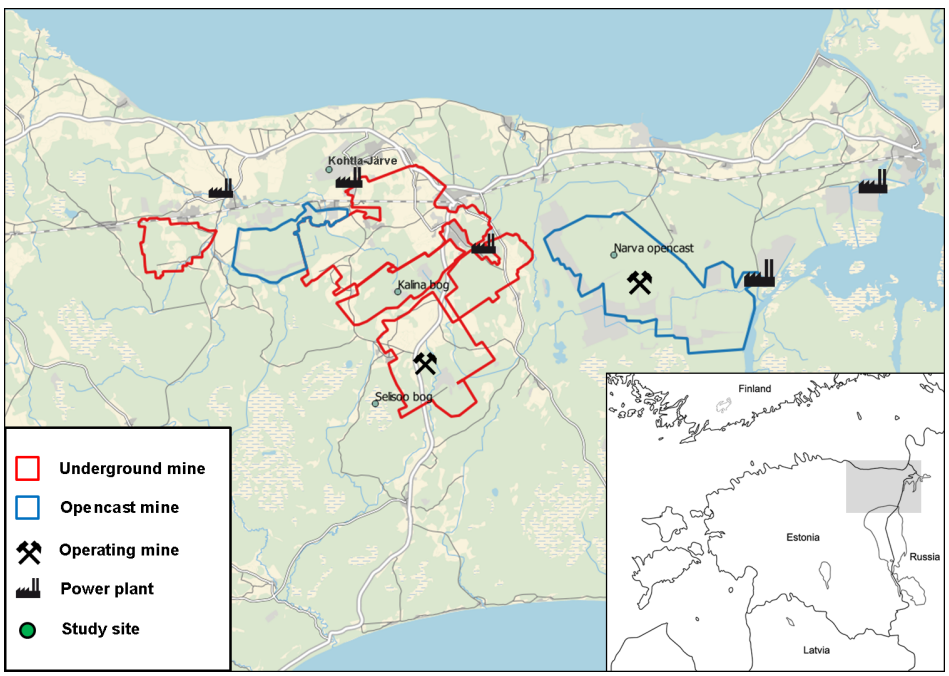

Figure 1. Location of the study sites in North-East Estonia.

Studies I and II were carried out in the Narva opencast oil shale mine (59°15'–59°18'N, 27°40'–27°46'E). Narva is the largest oil shale opencast mine with the area of approximately 100 km^2. Mining practices applied in the Narva mine are described in Pensa et al. (2004b) and Väli et al. (2008). Total concentrations of major soil nutrients (nitrogen, phosphorus, potassium) and soil pH_{KCl} under different-aged pine stands in the Narva mine are shown in Fig. 2. As the mine spoil is highly coarse, plants growing on spoil may occasionally suffer from drought, but the moisture regime in the top layers of levelled spoil is tolerable for most temperate tree species (Raid 1972).

Measurements were conducted in 80 circular plots of 100 m^2 in July 2002–2005. The plots represented eight site types (10 plots per site type): (*i*) pioneer stage on recently (~5 years ago) levelled mine spoil without trees; three series of planted Scots pine stands that were (*ii*) 10, (*iii*) 20, and (*iv*) 30 years old, respectively; nearly 30-year-old planted stands dominated by (*v*) larch (*Larix sibirica* Ledeb. and *Larix decidua* Mill.), (*vi*) silver birch (*Betula pendula* Roth), and (*vii*) alder (*Alnus glutinosa* (L.) Gaertn. and *Alnus incana* (L.) Moench.); and (*viii*) spontaneously developed stands in an area, where the mine spoil had been left unplanted (natural stands). All plots had the same initial distance from the adjacent unmined area, i.e. an equal dispersal distance for colonizing plants (Fig.1 in **I**). The Scots pine chronosequence (the first four site types) represents a successional trajectory that is typical of the study area. Thus, three Scots pine stands were selected for more detailed analysis of C stocks (Table 2 in **II**).

Study III was conducted in the Kohtla-Järve semicoke landfill (59°23′N, 27°13′E). The semicoke dumping site in Kohtla-Järve occupies 2.2 km^2 in total. However, the study area covered only the older part of the site, where semicoke was dumped from the mid-1930s until the mid-1990s. This old dumping area covered 1.3 km^2 and contained seven semicoke hills grown together. The ridge of the hills was more than 100 m long and the hillside slopes reached up to 60 degrees. The ridges were directed from east to west, so that hillsides were open to the north and south. As a growth substrate for plants, semicoke is a relatively poor material with low concentrations of nitrogen (N) and phosphorus (P), but with a relatively high concentration of potassium (K) (Kaar 2003). The topmost layer of semicoke is mostly coarse-grained and prone to drought. However, at a depth of 5–30 cm, the material is fine-grained and its water holding capacity is significantly higher. The pH of fresh semicoke is more than 12, but over time the pH of the topmost layers declines (Ostonen et al. 2006). The vegetated areas of the semicoke dumps were first determined on aerial photos issued by the Estonian Land Board. These vegetated areas were studied in June and July of 2004 and 2006; in total 76 plots of 10 × 10 m were established in the study area, distributed evenly across the vegetated area.

The study sites for **paper IV** included two bogs from North-East Estonia – Kalina and Selisoo, and a reference site Laeva located in central Estonia. The study was conducted in forested bog margins; two measurement plots of 300 m^2 were established in each site in 2009. The study sites located in North-East Estonia experienced a wide spectrum of anthropogenic disturbances:

- The most strongly disturbed site, Kalina, is located in the vicinity of a peat extraction field and its hydrological regime has also been affected by underground mining of oil shale. Kalina bog is also closest to the power plants that emitted alkaline air pollution until the 1990s, when the plants were equipped with effective dust filters. The long-term (from 1960 to 1997) daily deposition of alkaline fly ash varied between 50 and 350 mg m^{-2} day^{-1} in Kalina (Liblik et al. 2003). Two sample plots, denoted as K1 and K2, are separated

by a road that creates a difference in the hydrological regimes of the plots (Table 1 in **IV**).

- Two sites were studied in Selisoo bog. Seli1 (plots S1.1 and S1.2) is located in a former block peat cutting area, which has recovered by natural means. Seli2 (plots S2.1 and S2.2) is not directly affected by human activities, but it is drained by a natural stream. The long-term deposition of alkaline fly ash was 20–100 mg m^{-2} day^{-1} in Selisoo in 1960–1997 (Liblik et al. 2003). Selisoo bog is currently protected as a Natura 2000 area.
- Laeva site (plots L1 and L2) is located in Alam-Pedja Nature Reserve, which is a Natura 2000 and Ramsar site (no. 905). It is not affected by human activities and is not drained by ditches or natural streams.

2.2. Vegetation sampling and biomass estimates

Stand basal area (BA) at breast height, stand density and species composition were recorded in 100 m^2 subplots (papers **I, II, IV**). The age of trees was detected by coring stems at the height of 1.3 m above the ground. Stand height was measured with a precision of 0.5 m using a Haglöf electronic clinometer (Haglöf Sweden AB, Sweden) (**II, IV**). In bog sites (**IV**), trees were first divided into diameter classes according to their diameter at breast height (DBH), and the height of 2–3 stems in each diameter class was recorded. The stem radial growth rate was determined from increment cores (at least two cores per diameter class) taken at the height of 1.3 m (**IV**). Tree ring width and count (tree age) was measured with a WinDENDRO scanning and image analyzing device (Regent Instruments, Inc., Canada).

To estimate the biomass of Scots pines growing on the mine spoil (**II**), nine trees (three from each detailed-study stand) were felled for destructive sampling in summer 2004. Allometric equations to estimate dry mass of different tree compartments were derived using sample tree data (details are given in **II**). These empirical models were then applied to all Scots pine trees. Stem biomass for other tree species was estimated using Ozolinš's equation for calculating stem volume and wood densities (Lemming 2000; de Vries et al. 2003). Ground-area based estimates for different tree biomass compartments were obtained by dividing summed dry masses of all trees in a given plot by the plot area. Density and the biomass of the shrub layer, which included shrubs, tree saplings (height < 1.3 m), and *Rubus idaeus* L., were determined in two randomly placed 2 × 2 m quadrats (**I, II**). Within the quadrats, the total number of shoots was counted. Subsamples of different tissues were collected and dried at 65°C for 48 h to calculate dry masses (**II**). The carbon content of all biomass fractions was considered to be 50%.

In bog sites (paper **IV**), stem volumes (V) for Scots pines, silver birches and Norway spruces were calculated using functions from Laasasenaho (1982). V of other tree species was estimated according to the national forest management guide (Metsa ... 2009). For Scots pine, DBH versus V curves based on sample

trees were used to calculate the volume of all pine trees in the stand. Tree layer C stock, which included aboveground biomass, stumps and coarse roots (∅ > 1 cm), was calculated for plots using Eq. 1 (Minkkinen et al. 2001).

$$y = 0.048x^{0.949}, \quad (1)$$

where y is the C stock of trees (kg m^{-2}) and x is stand volume (m^3 ha^{-1}). Tree DBHs in previous year (2008) were obtained by subtracting the diameter increment measured from the increment cores from the 2009 DBH values. Annual stand volume and biomass increments were calculated by subtracting the values of 2008 from those of 2009. Two to three boxes (area 1 m^2) for collecting tree litter were installed in the plots in autumn 2008. The litter boxes were emptied once a month and the collected litter scores were pooled to calculate an estimate of annual litter productivity.

Samples of herb layer and dwarf shrubs were collected from five 0.4 × 0.5 m quadrats distributed evenly within a sample plot (**I, II, IV**). Within the quadrats, all shoots were cut at the ground level and sorted into species. Sorted samples were dried for 48 h at 60°C and weighed. The biomass of herbaceous species was used as a direct estimate of net productivity of the herb layer (**IV**). To estimate the net productivity of dwarf shrubs (**IV**), subsamples of each species and the fractions of shoots formed in a given year were measured separately. The total biomass of each species was then calculated to estimate the net productivity of dwarf shrubs. Nomenclature of vascular plant species in all studies followed Kukk (1999). Typical bog species were identified using species lists compiled by Kask (1982) for vascular plants and by Kannukene and Kask (1982) for bryophytes.

Moss productivity in paper **IV** was measured by applying the wire quadrat method (Ilomets 1974). In each plot, three wire quadrats of 10 × 10 cm were installed on the top of the moss layer in spring 2009. In December of 2009, the parts of mosses that had grown over the horizontal wires of the quadrats were cut, dried for 48 h at 65°C and weighed. The obtained values were multiplied by moss coverage estimated from five 0.4 × 0.5 m quadrats. The result was used as an estimate of the net productivity of the moss layer. Moss species growing in quadrates were identified; nomenclature of bryophytes followed Ingerpuu et al. (1998).

In paper **III**, the species composition in sample plots was recorded. The origin of the tree layer (either planted or natural) was established based on the information found from old reclamation projects. In the herb layer, the general coverage of vegetation was evaluated. When the herbaceous layer was very poor and sparse, the occurrence of species was recorded across the whole plot; in the cases the herbaceous layer was prosperous, quadrates of 1 × 1 m were used to count plant species. The number of quadrates within a sample plot ranged from two to five depending on the heterogeneity of the herbaceous layer.

2.3. Soil chemical analysis and estimation of SOC stocks

For chemical analysis, three samples of soil, either mine spoil in younger forest sites, a mixture of organic soil and mine spoil in older sites, or peat in bog sites, per plot were taken from a depth of 0–15 cm (**I, IV**). Routine chemical analyses were carried out at the Plant Biochemistry Laboratory of the Estonian University of Life Sciences. Total C content in peat samples (**IV**) was determined using the dry combustion method on a varioMAX CNS elemental analyzer (Elementar Analysensysteme GmbH, Germany) at the Laboratory of the Department of Soil Science and Agrochemistry, Estonian University of Life Sciences.

In sample plots established on mine spoil and semicoke dumps (**I** and **III**), the total thickness of the topsoil, consisting of the organic soil horizons (forest floor) and A-horizon, was measured in three places in each plot. Soil organic C stocks were estimated more accurately in three Scots pine stands growing on mine spoil (**II**). Three soil profiles were sampled in each detailed-study stand in July, 2005. To sample the forest floor, 0.4 × 0.5 m frames were used. Mineral soil was divided into A and C horizons (i.e. mine spoil with and without visible organic content, respectively) and sampled underneath the removed forest floor. Since the soils were very stony, soil samples were taken by excavation and sand replacement method. The thickness of the A horizon was measured in each pit, and the C horizon was sampled to a depth of 10 cm below the A horizon. The collected samples were air-dried and weighed. Mineral soil was sieved through a 2 mm mesh. The samples from each horizon were composited into a single sample per plot and ground before analyses. Estimation of SOC accumulation was obfuscated by the presence of carbon-containing oil shale fragments in the soil. The contribution of recent C to total SOC was determined by the radiocarbon technique. Radiocarbon dating of the samples was carried out at the AMS C^{14}-Laboratory of the University of Erlagen-Nürnberg, Germany.

The radiocarbon technique is based on the difference between ^{14}C activities of fossil and recent SOC. Since oil shale and other fossil fuels are composed of very old C, they consist of stable isotopes. Living plants, on the other hand, take up ^{14}C from the atmosphere and deposit it into the soil upon death. Thermonuclear bomb testing in the 1950s and early 1960s caused a substantial increase in atmospheric ^{14}C content, its peak levels reaching over 180 percent modern carbon (pMC) compared with 100 pMC, the hypothetical level in 1950. Since the banning of atmospheric nuclear tests in 1963, ^{14}C activity in the atmosphere has dropped and levelled out. Thus, the ^{14}C activity of recent SOC depends on the amount of recent carbon and the time period it entered soil, and was estimated by combining a simple model of litter production and the record of atmospheric ^{14}C. The relative proportion of recent C was then obtained by mass balance using the measured ^{14}C activity in soil sample. A detailed description of the C stock calculations is given in paper **II.**

2.4. Measurements of peat respiration rates and methane emissions in bog margins

CO_2 emissions from peat or peat respiration rates (R_s) in paper **IV** were measured using the soda lime method (Edwards 1982; Grogan 1998). At each site, R_s rates were measured monthly from April to December 2009 using the methodology described by Keith and Wong (2006). Seven marked chambers (five measurement chambers and two blank chambers) with a surface area of 0.055 m^2 were installed in each experimental plot during the previous autumn. Inside the measurement chambers, all aboveground plant biomass was removed including the green parts of mosses. Thirty grams of soda lime was kept in each chamber, in both measurement and blank chambers, for at least 24 h. During that period, chambers were kept closed with a sealed lid and a polyethylene bag (sealed with an elastic band). R_s (g C m^{-2} day^{-1}) was calculated according to Keith and Wong (2006).

Measurements of methane (**IV**) were carried out from July to December simultaneously with the soda-lime measurements using a portable FTIR gas analyzer (Gasmet DX4030; Gasmet Technologies Oy, Finland). For the FTIR measurements, a transparent polypropylene chamber with a volume of 0.02 m^3 was used. Vegetation was not removed from the chamber. In each sample plot, methane emissions were measured in two places.

Water table depth (WTD) and peat temperature at a depth of 10 cm were also recorded at the same time as the measurements of peat C gas fluxes. In addition, two peat cores per plot were taken to visually estimate whether peat decomposition and changes in vegetation composition had occurred over time.

2.5. Data analysis

Changes in *V* and soil properties during succession on mine spoil afforested with Scots pine were described using Spearman rank order correlations coefficients (Spearman r_s). The effect of site type on the stand structural characteristics and soil properties was estimated by general linear model (GLM), where predicting factors were site type and stand age; the latter was significant only in the case of soil [P]. Pairwise comparisons were done with Tukey's HSD test; the homogeneity of variances was tested with Levene's test. When necessary, variables were log-transformed to improve the homogeneity of variances.

To describe the patterns in species composition on mine spoil, semicoke dumps and bog margins, nonmetric multidimensional scaling (NMDS) was used. The input data were relative biomasses of herbaceous species (**I**) or presence of species in a study plot (**III** and **IV**). Herbaceous species biomass and richness in paper **I** were expressed at plot and site type levels. The change in species composition within a site type (β diversity) was expressed as the ratio of site-level species richness (γ diversity) to the mean number of species in a plot (α diversity). Differences in the herb layer α diversity between site types were

assessed by one-way analysis of variance (ANOVA). The variation in the composition of herbaceous vegetation among site types established on mine spoil was estimated using a permutation-based nonparametric multivariate ANOVA (MANOVA; Anderson 2001; McCune and Grace 2002). Before the analysis, biomass values were relativized (i.e., a biomass of a species in a given plot was expressed relative to the total biomass of that species across all plots). Site type was used as a fixed factor in the MANOVA. To find which species could be associated with a particular site type, indicator species analysis was employed (McCune and Grace 2002). The indicator value shows whether the frequency and abundance of a species in a particular site type is higher than expected by chance. The differences in the abundance of Grime's (1979) competitive, stress-tolerant, and ruderal plant strategies were compared among site types by using the functional signature approach suggested by Hunt et al. (2004).

Linear mixed model using REML estimation of parameters was applied to peat R_s data to test the effect of sites (categorical fixed factor) and time (continuous fixed factor) on respiration rates in different chambers nested within plots (categorical random factors). To stabilize the variance among sites, the response variable was log-transformed. For the same purpose, outliers (one per measurement day per site) were removed from the dataset. Model building followed the steps suggested by Pinheiro and Bates (2000). The final optimal model was a random intercept model containing site, time and a quadratic form of time, along with two-way interactions between site and the two time variables (Eq. 2).

$$(\log R_s)_{ijk} = \alpha + Site_{ijk} \times Time_{ijk} + Site_{ijk} \times Time^2_{ijk} + a_k + a_{(j|k)} + \varepsilon_{ijk} \quad (2)$$

where $(\log R_s)_{ijk}$ is the $\log_{10}$ value of peat respiration for the ith observation in the jth chamber in the kth plot. The term a_k is a random intercept and allows for variation between plots, $a_{j|k}$ represents the variation between chambers in the same plot, ε is a random error term, and α is the overall constant population mean of peat respiration.

Relationships between soil C emissions and WTD in bogs were estimated using Spearman rank order correlation. Plot-level median R_s for the growing season and median CH_4 flux rates were related to WTD by nonlinear regression procedure. Pearson correlation coefficients (r) were used to describe relationships between environmental and stand variables. ANOVA with Tukey's HSD test was applied to compare soil chemical parameters and productivity between sites, and Kruskal–Wallis rank sum test was applied for gas flux data which were not normally distributed. Post-hoc multiple comparisons of emission rates between sites were performed as described in Siegel and Castellan (1988). All statistical tests were done at a significance level of $\alpha = 0.05$.

3. RESULTS AND DISCUSSION

3.1. Tree growth and soil development in a Scots pine chronosequence

Since more than 80% of the exhausted oil shale mines in Estonia have been afforested with Scots pine (Kaar 2010), pine chronosequence represents a successional trajectory that is typical of the mined-out areas. Successional changes in stand structure, soil properties and species composition were studied using a space-for-time substitution that is considered the most feasible option to investigate long-term changes in forests over the short term (Powers and Van Cleve 1991; Peichl and Arain 2006). The chronosequence approach is based on the assumption that sampled stands of different age represent the developmental stages of a forest. Indeed, all the studied stands were initiated from the same pioneer stage – bare mine soil, although in the Narva opencast the substrate is very heterogeneous (Vaus 1970; Kuznetsova and Mandre 2006). We also found that soil nutrient concentration varied greatly among young stands (Fig. 2D–F). Also, the survival and growth rate of trees was not uniform across the sites, as the growing stock ranged from 100 to 360 m^3 ha^{-1} among 30-year-old Scots pine stands (Fig. 2A). Nevertheless, as our chronosequence consisted of 40 stands, the development of Scots pine stands in the Narva oil shale mine should be represented adequately. Three plots where carbon pools were studied in detail were situated in close proximity and the relationships between stand age and measured parameters of these stands were near to the average of all sampled plantations (Fig. 2).

Of all measured parameters, stand growing stock and topsoil (O and A horizons) depth changed most strongly with stand age (Fig. 2) and were also mutually correlated (Spearman $r_s = 0.94$; $p < 0.05$). Fig. 2A demonstrates that despite seemingly harsh growth conditions, Scots pine shows very good growth performance on the mining substrate (see also Fig. 2 in **II)**. Previous studies (Kaar and Raid 1992; Kuznetsova et al. 2011; Pietrzykowski and Socha 2011) have also found that Scots pine is well adapted to calcareous mine spoil conditions and the growth rate of pine plantations established in exhausted oil shale opencasts may even exceed the average growth rate of Estonian forests in the most productive site types (Kiviste et al. 2010).

Organic-rich topsoil started to develop above mine spoil during the second decade of the stand growth, reaching up to 9.5 cm in 30-year-old stands (Fig. 2B). Also, remarkable changes in soil chemical properties occurred during the pedogenesis: total [N] in mine soils significantly increased with stand age, whereas pH, total [K], and especially total [P] significantly decreased (Fig. 2C-F). N is deposited into soil through accumulation of organic matter and fixation of atmospheric N_2, and therefore increases during primary succession; P is released through weathering of minerals, which is a slow process. A decrease in available P in mine spoil may result from increasing uptake and assimilation of P by plants during stand development (Šourková et al. 2005). The same probab-

ly also applies to K. In addition, K may be lost from soil through leaching. Soil texture improved in the course of soil development, as the coarse fraction content and soil bulk density decreased and specific surface area increased with stand age in three intensively studied Scots pine stands (Karu 2005). Similar changes in mine spoil chemistry and texture during succession have also been observed in other studies (Akala and Lal 2000; Reintam 2001, 2004; Bodlák et al. 2012). The changes accompanying SOC accumulation are beneficial for plant productivity, as forest growth on permanent inventory plots established on oil shale opencasts is accelerating compared to the average growth in Estonian forests growing on fertile sites (Korjus et al. 2007; Kiviste et al. 2010).

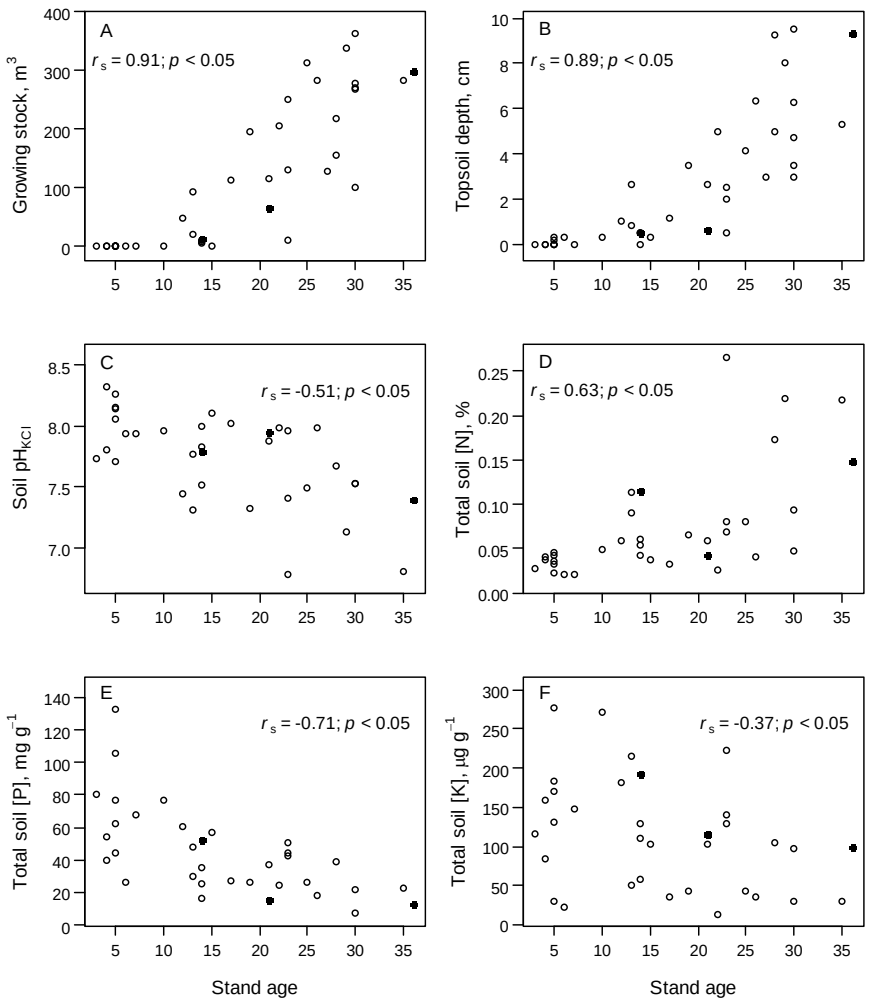

Figure 2. Change in growing stock and soil properties (as plot-level average) in a Scots pine chronosequence. Filled points denote plots selected for investigating carbon sequestration and r_s stands for Spearman rank order correlations coefficient.

3.2. Comparison of productivity and soil properties between 30-year-old plantations

Compared to Scots pine stands, 30-year-old silver birch, black alder and larch stands had higher growing stocks (Fig. 3A) and deeper topsoil (Fig. 3B), although differences were not significant due to the high variability within stand types. This is consistent with other researchers reporting that these species have higher growth rates on oil shale mine spoil (Kaar and Raid 1992; Kaar 2002b; Lõhmus et al. 2007; Kiviste et al. 2010; Kuznetsova et al. 2011). We also found that alders had the lowest variability in growing stock (Fig. 3A), which probably results from their better survival compared to silver birch and Scots pine (Kuznetsova et al. 2011) and its suitability for stony substrates (Kaar and Raid 1992; Kaar 2002b; Kaar 2010). Larch species have not been used for reclamation of exhausted oil shale opencasts since 1994, as it is not recommended (Kaar 2010). On the other hand, more extensive planting of deciduous species has been recommended for both economic and ecological reasons (Kaar 2002a, b; Pensa et al. 2004b; Kuznetsova et al. 2010).

Black alder and silver birch improve their mineral nutrition on mine spoil by supporting rhizosphere microbial communities and through high morphological plasticity of fine roots (Vares et al. 2004; Ostonen et al. 2006; Lõhmus et al. 2006, 2007; Rosenvald et al. 2011). In addition, alders form symbiotic associations with actinomycetes from genus *Frankia* that are able to fix atmospheric nitrogen (Huss-Danell 1997). The nutrient-rich and easily degradable litter of deciduous trees, especially N_2-fixing species, also enhances soil development on mine spoil (Kaar and Raid 1992; Frouz et al. 2009; Frouz et al. 2013; Šnajdr et al. 2013).

Although there were no apparent differences in soil pH and nutrient concentrations under very young (1–7 years old) Scots pine, black alder and silver birch stands growing on oil shale mine spoil (Kuznetsova et al. 2011), in 30-year-old plantations the effect of tree species on soil properties was evident. In our sites, black alder demonstrated the most positive effect on soil, as soil pH was lowest under alder stands, and [N] and [P] were highest. Similarly, Vares et al. (2004) found that N and P pools in the mine soil were approximately 2-fold higher in 21-year-old alder stand than in adjacent Scots pine plantations of the same age. The rate at which nitrogen compounds are transformed to forms available for plants is an important determinant of herbaceous biomass and species composition in terrestrial ecosystems (Bengtson et al. 2006). Thus, because alder stands had higher soil [N] and [P] compared to other 30-year-old sites, they also permitted the most productive herb layer (Table 1), which is an extremely important factor for humus formation on mine spoil (Reintam et al. 2002). Also, on semicoke dumps the thickest topsoil and highest coverage of herb layer were observed under alder plantations (Table 1 in **III**).

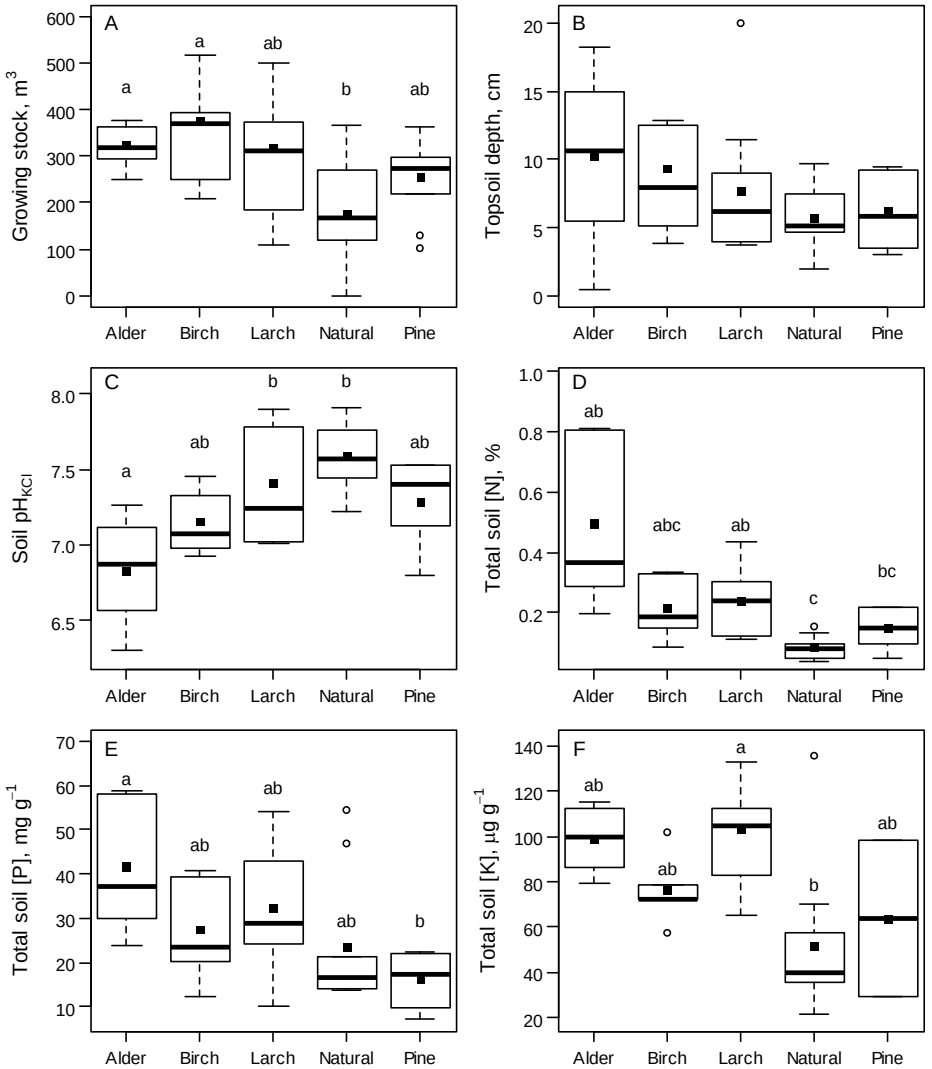

Figure 3. Growing stock and soil properties in 30-year-old Silver birch, black alder, larch and Scots pine stands and naturally recovered sites. Letters denote significant differences among the groups. Soil [N] was log-transformed before the analysis. The box represents lower and upper quartiles and whiskers, range without outliers. Line – median, square – mean, points – outliers.

Table 1. Dominant species and structural characteristics of vegetation in different stand types on oil shale mine spoil. Letters denote significant differences between groups.

Site type	Species no	Planted species	Dominant species (relative abundances)	Structural characteristics
Tree layer				Average basal area ± SD (m^2 ha^{-1})
Pioneer	–	*Pinus sylvestris*	–	0
Pine 10	3	*P. sylvestris*	*P. sylvestris* (91)	9.04 ± 9.33 a
Pine 20	5	*P. sylvestris*		28.3 ± 11.3 bc
Pine 30	6	*P. sylvestris*		36.5 ± 9.2 bc
Natural	7		*Betula pendula* (52), *Salix* spp. (22)	22.1 ± 15.7 ab
Birch	5	*B. pendula*	*B. pendula* (92)	42.2 ± 13.3 c
Larch	7	*Larix* spp., *B. pendula*, *Tilia cordata*, *Acer platanoides*	*Larix* spp. (95)	35.4 ± 11.9 bc
Alder	5	*Alnus glutinosa*, *A. incana*	*A. glutinosa* (61), *A. incana* (19), *Alnus* spp. (12)	38.7 ± 5.1 c
Undestorey				Shoot density ± SD (shoots 4 m^{-2})
Pioneer	3		*Salix* spp. (67), *B. pendula* (31)	7.2 ± 8.4 ab
Pine 10	7		*Salix* spp. (70)	4.0 ± 3.3 a
Pine 20	6		*Salix* spp. (32), *B. pendula* (24)	3.4 ± 3.0 a
Pine 30	10		*B. pendula* (44), *Rubus idaeus* (22)	5.2 ± 5.0 ab
Natural	13		*B. pendula* (65)	11.2 ± 14.3 ab
Birch	11		*R. idaeus* (62), *B. pendula* (22)	15.6 ± 8.6 b
Larch	10		*R. idaeus* (49), *B. pendula* (26)	6.8 ± 5.6 ab
Alder	8		*R. idaeus* (94)	14.6 ± 7.0 b

Table 1. (continued).

Site type	Species no	Planted species	Dominant species (relative abundances)	Structural characteristics
Herb layer				Average biomass ± SD (g m^{-2})
Pioneer	31		*Calamagrostis epigeios* (58), *Tussilago farfara* (32)	49.8 ± 27.1 ab
Pine 10	28		*T. farfara* (71)	30.7 ± 42.8 ab
Pine 20	25		*C. epigeios* (24), *C. arundinacea* (14), *T. farfara* (10)	15.5 ± 14.3 ab
Pine 30	28		*Orthilia secunda* (25), *C. arundinacea* (24), *Fragaria fesca* (14)	21.3 ± 14.9 ab
Natural	50		*C. arundinacea* (22), *R.saxatilis* (13), *Cirsium oleraceum* (11)	17.3 ± 11.7 ab
Birch	27		*C. arundinacea* (63)	36.4 ± 28.3 ab
Larch	33		*F. fesca* (30), *C. arundinacea* (23)	13.8 ± 8.6 a
Alder	36		*Urtica dioica* (16), *C.arundinacea* (13)	78.7 ± 46.1 b

3.3. Carbon sequestration by Scots pine stands

Based on the results of our studies (**I, II**) and previous research, it can be concluded that the productivity of forests and soils recovers fast in afforested areas of oil shale mine spoil, although it depends on species and substrate quality. Due to the better drainage of mine spoil compared to pre-mining soils, timber yield in afforested oil shale opencast areas is even higher than in forests existing before mining started (Reintam and Kaar 1999; Danilov 2010). However, boreal and hemiboreal forests on natural soils store more C in soil than in vegetation (Dixon et al. 1994; Mahli et al. 1999; Pan et al. 2011) and thus the potential for SOC accumulation in oil shale mining spoil could also be substantial. Therefore, three Scots pine stands (Table 2 in **II**) were studied in more detail in order to estimate the rate of carbon accumulation and to determine the distribution of sequestered carbon along forest ecosystem compartments.

In the aboveground compartment, the share of pine biomass was highest and increased with age: 14 years after afforestation pine biomass comprised 90% of total aboveground biomass, in 21-year-old stand the share of pines was 98% and in 35-year-old stand 99% (Table 2). The proportions of shrub and herb layers were largest in the youngest stand (7.8% and 1.8%, respectively) and decreased during stand development. In the oldest stand, shrubs and herbs formed less than 1% of the total vegetation C pool. Compared to our sites, Scots pine stands established on mine spoil in Poland had considerably lower aboveground C stocks despite more favourable climatic conditions (Pietrzykowski and Daniels 2014). Stem biomass exhibited the largest portion of the total tree biomass in all three plantations (Table 2). Its contribution increased with age, but 21 years after stand establishment, stem biomass still accounted for only 54% of total tree biomass. This demonstrates that ignoring other biomass pools in forest C stock estimations may considerably underestimate the tree biomass in young stands, as was also pointed out by Peichl and Arain (2006).

Belowground biomass estimates in the detailed-study stands included only stumps and coarse roots of pines and were therefore underestimated. Nevertheless, other studies on Scots pine have shown that most of the belowground biomass resides in stumps and coarse roots, and the share of fine roots from the total tree biomass in 10- to 50-year-old stands varies from <1% to 15% (Vanninen et al. 1996; Ilvesniemi and Liu 2001; Helmisaari et al. 2002; Xiao and Ceulemans 2004) and decreases with age (Vanninen et al. 1996; Helmisaari et al. 2002). Measured values of fine root biomasses in young Scots pine forests range from 1.5 to 4 t ha^{-1} (up to <2 t C ha^{-1}) (Makkonen and Helmisaari 1998; Oleksyn et al. 1999; Vanninen and Mäkelä 1999; Helmisaari et al. 2002).

Since soils in restored oil shale opencasts also contain fragments of carbon-containing oil shale, the measurements of soil total organic carbon (TOC) overestimate real carbon sequestration in the ecosystem. Several methods have been used to differentiate between plant-derived recent carbon and fossil carbon in coal-, lignite- and oil shale mines (Ussiri et al. 2014), although no standard methodology has been worked out. For our sites, we used a radiocarbon

technique that was proved to be effective in previous works (Rumpel et al. 1999, 2003; Morgenroth et al. 2004; Fettweis et al. 2005).

The calculated ^{14}C activities of recent SOC were 107.9, 108.7, and 110.8 pMC for the first, second and third stand studied in detail, respectively. Samples taken from the forest floor horizons had practically the same activities (Table 5 in **II**), supporting our approach, as forest floor contains only plant-derived C. The share of recent C decreased with soil depth – SOC in the upper 10 cm layer of the C horizon (mine spoil) originated mostly from oil shale, but also contained considerable amounts of recent carbon (16–39% of TOC, 0.4–1.4% of soil fine fraction). As part of this C may be a legacy from pre-mining ecosystems, recent C stocks in the C horizon were omitted from the estimations of total soil and ecosystem C sequestration, which is therefore somewhat under-estimated.

The average annual increment of soil C (i.e. recent SOC stock in topsoil divided by stand age) increased with stand age in accordance with Vindušková and Frouz (2013), who found that SOC accumulation under coniferous forests on opencast coal and oil shale mines reaches maximum rates 30–40 years after afforestation. According to their meta-analysis, average SOC accumulation rate in young (<30 years old) coniferous forests growing on coal and oil shale mining spoil is 0.81 ± 0.38 t ha^{-1} yr^{-1} (Vindušková and Frouz 2013). We obtained a similar value in the oldest detailed-study stand, where the average rate of SOC sequestration was 0.33 t ha^{-1} yr^{-1} in the forest floor and 0.51 t ha^{-1} yr^{-1} in the A horizon. These values are within the range of rates reported by Reintam et al. (2002), who studied soil formation under 29–34-year-old Scots pine plantations growing in the same mine. They estimated carbon sequestration up to 1.74 t ha^{-1} yr^{-1} for the forest floor and humus horizon combined, and 2.38 t ha^{-1} yr^{-1} for the whole soil profile, including AC horizon. Compared with our results, the rate of SOC sequestration is higher in a lignite mine in Germany afforested with Scots pine, in which 50 t ha^{-1} new C was added to the soil in less than 40 years (Rumpel et al. 1999; Fettweis et al. 2005).

Carbon storage may be considered one of the indications of ecosystem recovery in post-mining sites (Frouz et al. 2009). Scots pine plantations in the Estonian oil shale opencasts accumulated >130 t C ha^{-1} in less than 40 years after establishment (Table 2). As expected, tree stems contributed most to the carbon sequestration – their share increased with age from 28% to 51% (Fig. 4 in **II**). Recent SOC formed 5% of the total ecosystem carbon stock in the youngest, 15% in the medium and 23% in the oldest stand. These results are comparable to the findings of Frouz et al. (2013) for pine forests on a colliery spoil heap.

Our results suggest that soils contribute significantly to C accumulation during early forest succession on degraded lands. However, because of the low number of study sites and high variability of the parent substrate, there still remain uncertainties concerning the average rate of SOC accumulation in reclaimed oil shale opencasts and its dependence on environmental factors and site characteristics. Future studies should cover a wider range of degraded mine

lands and also include deciduous tree species to investigate more thoroughly our previous finding that deciduous species enhance soil development on oil shale mining spoil.

Table 2. Plant-derived recent carbon stocks in detailed-study stands of the Scots pine chronosequence on oil shale mine spoil.

Fraction	Carbon stock (t C ha^{-1})		
	Stand 1 (age 14/15 years)	**Stand 2** (age 21/22 years)	**Stand 3** (age 35/36 years)
Stems	2.15	15.68	68.21
Needles	1.96	3.32	4.44
Living branches	1.71	4.0	9.30
Dead branches	0.04	2.66	8.07
Willows in tree layer	0.02	---	---
Shrub layer (mean ± SD)	0.51 ± 0.61	0.58 ± 0.65	0.45 ± 0.55
Herb layer (mean ± SD)	0.11 ± 0.06	0.01 ± 0.01	0.04 ± 0.02
Total aboveground C	**6.50**	**26.24**	**90.51**
Stumps and coarse roots	0.90	3.28	11.63
Recent C in O-horizon (low estimate – high estimate)	0.13 (0.06 – 0.27)	4.58 (2.17 – 9.65)	12.21 (5.80 – 25.74)
Recent C in A-horizon (low estimate – high estimate)	0.24 (0.12 – 0.51)	0.44 (0.21 – 0.92)	19.04 (9.04 – 40.14)
Total recent C	**7.78**	**34.53**	**133.40**

3.4. Successional changes in species composition on oil shale mine spoil and semicoke dumps

Besides the recovery of ecosystem functions in terms of C sequestration and provision of biomass in oil shale opencasts, changes in species composition and diversity were also assessed in both the Narva mine and Kohtla-Järve semicoke dumps. During the first 35 years of succession, the number of species encountered in the tree layer and understory in Scots pine chronosequence plots established in opencast increased slowly (Table 1). The most common invaders in the tree layer were willows and silver birch, the same species that also dominated the understory of the pine stands and were the most successful woody species in unreclaimed areas (Table 1).

Species diversity in the ground vegetation of pine forests did not change significantly with stand age (Table 2 in **I**). However, we found significant diffe-

rences in the species composition of the herb layer among stands with different ages (permutation-based MANOVA, $F[3,36] = 1.62$, $p < 0.001$). Further analysis of the data on species composition using NMDS ordination resulted in a three-dimensional solution (final stress 18.5, final instability <0.001, 250 randomized runs in Monte Carlo test, $p = 0.004$; Fig. 3A in **I**). Cumulatively, the NMDS axes explained 41% of the variance in species composition. The first axis of the chronosequence NMDS plot ($R^2 = 0.14$) correlated positively with topsoil depth, tree biomass, age, and soil [N], but negatively with the incident solar radiation, soil pH, [P], and [K] (Table 4 in **I**), thus corresponding to the changes occurring with stand age. The second axis ($R^2 = 0.13$) gave similar but weaker correlations with site characteristics. Change in light conditions for ground vegetation in the course of succession is a well-known cause for species alternation (Okland 2000; Uotila et al. 2005) and is probably the main reason for the alignment of stands along the first axis (Fig. 3a in **I**). Accumulation of the forest floor may be another important factor causing changes in species composition in young stands growing on mine spoil (Mudrák et al. 2010). The third NMDS axis ($R^2 = 0.14$) correlated most strongly with herbaceous biomass.

On semicoke dumps, the most common tree species was silver birch. Birch stands included both planted and spontaneously grown sites of different ages (Table 1 in **III**). Thus, successional changes in the ground vegetation were assessed using the birch chronosequence. Applying NMDS ordination on the species composition data resulted in a three-dimensional solution (final stress 10.0, final instability < 0.001, 250 randomized runs in the Monte Carlo test, $p < 0.01$). Cumulatively, the NMDS axes explained 94% of the variation in the species composition. The study plots of different age classes were homogeneously distributed across the NMDS space, indicating that the species composition of herbs did not differ among stands of different age (Fig. 2b in **III**). It seems that ecosystem development is slower on dumps than on oil shale mine spoil, as the average topsoil depth in ~30-year-old silver birch plantation was only 3 cm and the coverage of the herb layer was less than 50% (Table 1 in **III**). Therefore, successional changes in species composition also take more time compared to oil shale opencasts.

The herb layer during the pioneer stage in the oil shale opencast was dominated by *Calamagrostis epigeios* (L.) Roth and *Tussilago farfara* L. (Table 1). *T. farfara* was also dominant in the 10-year-old pine stands. Both species are light demanding, but tolerate lack of nutrients; they are anemochores and have good competitive abilities (Grime et al. 1988). Thus, they can easily occupy large areas within a short period. *C. epigeios* is a very successful colonizer also in Central-European coal mine spoils, often suppressing the establishment of other species (Hodačová and Prach 2003; Mudrák et al. 2010). *T. farfara* and *C. epigeios* were also among the most common species in young birch stands on semicoke hills, together with *Taraxacum officinale* F.H. Wigg (coll.) and *Achillea millefolium* L. (Table 2 in **III**). Unlike the Narva mine, where surrounding unmined land is dominated by mixed forests, semicoke dumps are in

the vicinity of urban and industrial areas, which probably have an effect on its species composition.

The development of afforested areas from open habitats to those dominated by trees was accompanied by a decline in herbaceous biomass and the replacement of light-demanding species by shade-tolerant ones. In the 30-year-old pine stands, *T. farfara* and *C. epigeios* made only a marginal contribution to the total herbaceous biomass, while *Calamagrostis arundinacea* (L.) Roth, *Fragaria vesca* L. and *Orthilia secunda* (L.) House were the dominant species (Table 5 in **I**). The latter had a significant indicator value for 30-year-old pine stands, showing its strong association to that particular stand type. The percentage of ancient forest species, i.e. plants associated with a long continuous forest history, for each site was found based on the species list given by Hermy et al. (1999). According to that, there were no ancient forest understory species in the pine plantations belonging to the three younger age classes, but in 30-year-old stands, their share in herbal biomass was already 18%. Similarly, Laarmann et al. (2015) found that the share of forest understory species increased with time in afforested Aidu oil shale mine. As the stands are still young, it would be important to continue the monitoring of vegetation development in these forests to learn whether and when the species composition typical of forest ecosystems will be restored.

Open nutrient-poor habitats prevailing in post-mining sites are thought to favour poor competitors characteristic of grasslands, such as orchids (Tischew and Kirmer 2007; Řehounková et al. 2011; Tischew et al. 2014). Indeed, on semicoke dumps we identified seven orchid species [*Dactylorhiza baltica* (Klinge) Orlova, *D. fuchsii* (Druce) Soó, *D. incarnata* (L.) Soó, *D. maculata* (L.) Soó, *Epipactis atrorubens* (Hoffm.) Besser, *E. helleborine* (L.) Crantz, and *Orchis militaris* L.], which is more than in the surrounding forests. One reason might be the high abundance of fungi, which are essential for the germination of orchid seeds. Another reason is probably the low abundance of competitive plant species, which might suppress the growth of orchids to be classified mainly as stress-tolerant species (Grime et al. 1988). This explains why only one orchid species was present in the *Alnus* stands, where the coverage of herbaceous vegetation was highest. Three orchid species (*E. atrorubens*, *E. helleborine* and *Orchis militaris*) were found in mine spoil plantations (Table A1 in **I**). However, none of the orchid species found on semicoke dumps and oil shale opencasts are rare or endangered in the Estonian flora (Kukk and Kull 2005), and thus the value of the studied areas as habitats for protected species is low.

3.5. The effect of planted tree species on the development of ground vegetation

Planted tree species had a strong effect on the development of the ground vegetation on reclaimed oil shale mine spoil, as species composition of the herb layer differed significantly among 30-year-old forest plantations and spontaneously recovering sites ($F[4,45] = 1.58$, $p < 0.001$). Vegetation of alder stands was most distinct from the others (Table 3B in **I**). The significant indicator species in birch stands was *C. arundinacea*, whereas *Lysimachia vulgaris* L. and *Elymus caninus* (L.) L. were indicators for natural stands. In alder stands, nine species had a significant indicator value (Table 5 in **I**). The number of the unique species was highest in natural stands and alder plantations, although they covered the smallest area of the compared site types and were surrounded by pine forests. This is strong evidence that the formation of the herbaceous vegetation in 30-year-old stands was affected primarily by recruitment limitations rather than by dispersal limitations. It is reasonable to assume that these species had to disperse through pine stands, but were unable to germinate or survive in the harsh environment.

The occurrence of *Urtica dioica* L., a nitrophyte, only in alder stands is an example how the engineering effect of tree species favours some species and sets recruitment limitations for others. Because alder stands had the highest soil [N] and [P], their herb layer was lush and included fast-growing competitors as dominant species. However, the competitors obviously suppressed slow-growing stress-tolerant species, such as *O. secunda*, which was abundant in 30-year-old pine stands. The dominance of stress-tolerant species in pine stands is attributable to slow litter decomposition rates and low soil nutrient concentrations, which are more characteristic of coniferous stands as than broadleaved or mixed forests (Saetre et al. 1997; Saetre 1999; Macdonald and Fenniak 2007; Frouz et al. 2013). Although competitors were able to disperse to pine stands (otherwise they could not colonize alder stands), poor soil conditions limited their survival in pine plantations. Thus, consistent with Mudrák et al. (2010) and Laarmann et al. (2015), we propose that the observed differences in the herbal layers of alder and 30-year-old pine stands are caused by the quality of litter through which tree species affect soil properties.

The greatest plant species diversity (especially γ and β diversities) was recorded in naturally recovered sites (Table 2 in **I**), as reported in several previous studies (Hodačová and Prach 2003; Pensa et al. 2004b; Mudrák et al. 2010). Spontaneous succession has been considered a viable alternative to reclamation of disturbed areas, where environmental conditions are not extreme, and where negative influences on the surroundings are not expected (Prach and Hobbs 2008). In the Narva mine, forest ecosystems can develop relatively fast through natural succession as evidenced by the lack of differences between growing stock and topsoil depth between 30-year-old natural stands and Scots pine plantations (Fig. 3A and B). Another successful example of the natural recovery of exhausted oil shale opencasts is the Küttejõu mine, which was closed before

the afforestation campaign started, and is now covered with a spontaneously developed diverse mixed forest (Pensa et al. 2004b). However, natural stands have lower economic value compared to commercial plantations (Luud and Pensa 2004), which has shaped a negative view on spontaneous recovery among foresters. On the other hand, in some areas of the Narva mine, reclamation would be very complicated (Viil 2010) or unreasonable, for instance, in the lower sites that will probably be flooded when the mine is closed (Kaar et al. 2010). Our study suggests that by letting natural succession occur in small areas within plantations, the vegetation and landscape in reclaimed opencasts could be diversified without remarkable economic costs.

3.6. Impact of anthropogenic activities on vegetation in two North-East Estonian bogs

Since bog ecosystems are sensitive to atmospheric nutrient influx, we expected species composition and vegetation structure in our North-East Estonian sites, historically affected by high alkaline air pollution load, to differ from the pristine bog located in Central Estonia (Laeva site). Analysis of species composition in peat cores showed that all the plots had similar *Sphagnum*-dominated vegetation in the past; therefore we assumed that all changes in vegetation were caused by anthropogenic activities. Alkaline fly ash levels have been highest at Kalina, which is the bog site closest to the great Estonian power plants and belonged in the zone of relatively high pollution level in 1960–1980 (Liblik et al. 2003). At the same time, Selisoo belonged in the zone of moderate air pollution. However, in both zones noticeable anthropogenic impact on the bog vegetation has been described, such as deterioration of the growth of *Sphagnum* mosses and changes in species composition more similar to that of fens (Liblik et al. 2003).

As expected, shifts in species composition had occurred at both Kalina and Selisoo bogs as revealed by the NMDS ordination (Fig. 2 in **IV**). Vegetation in Kalina plots, especially in K1, was most distinct from the others (Bray-Curtis dissimilarity index was 0.6 between K1 and Laeva plots). Species richness was also highest at Kalina: 24 species were found in K1 and 19 in K2. At the same time, the share of species typical of bog habitats was only 33 and 58%, respectively. In Kalina, we encountered several species that are adapted to more nutrient-rich forest ecosystems, including *Picea abies* (L.) H. Karst, *Fragaria vesca* L. etc. Vegetation at Laeva consisted entirely of bog species, and at the Selisoo sites such species formed 78–92% of the species list. Moss cover was sparse in K1 (coverage 23%), but still contained some sphagna (Table 4 in **IV**). On the other hand, in S1.2 the cover of mosses extended almost to 100%, but consisted of forest species. The most important environmental factors that determined species composition in the study plots were peat pH_{KCl} ($R^2 = 0.79$; $p = 0.038$), which reflects the impact of alkaline air pollution, C content of the

peat (R^2 = 0.82; p = 0.012), and average WTD (R^2 = 0.75; p = 0.042), which separated the plots along the second ordination axis (Fig. 2 in **IV**).

Differences in water level among the sample plots were relatively small, as average WTD ranged from 4.5 in Laeva plots up to 28.5 in S1.2 (Table 1 in **IV**). There was quite large within-site variation of WTD in drained Kalina and Seli1 sites; in the latter, the difference between two measurement plots of average WTD was as much as 20 cm. In plot K2 the WTD was only slightly deeper compared to the pristine bog, ranging from 0 to 14 cm.

Probably both alkaline pollution and lowering of the water table contributed to the increased tree productivity in our studied North-East Estonian bogs. At Kalina, trees were only up to ~50 years old (at breast height), which implies that the tree layer started to develop after the onset of the intense anthropogenic impact caused by the fast industrial development in North-East Estonia. Also, the growth rate of young bog pines has increased abruptly in the second half of the 20th century at Selisoo (Karu et al. 2014). Among our study plots, tree stand productivity both in absolute and relative terms was highest in plots S1.2 and K1 (Fig. 3 in **IV**), which were the most disturbed plots based on species composition. The increase in tree growth rates indicates a worsening of the growth conditions for *Sphagnum* mosses. Thus, we found that the productivity of mosses at the sites experiencing more severe human impact (Kalina and Seli1) was significantly lower compared to the undisturbed sites (Mann-Whitney's U test, $Z = -2.21$; p = 0.021). In the long term, it may lead to a shift in the functionality of bog ecosystems, i.e. they may shift from C sinks to C sources.

3.7. Peat respiration rates and methane emissions in Kalina and Selisoo and their dependence on environmental factors

When comparing the annual course of daily R_s between sites, it is evident that differences in respiration rates occurred during the peak growing season, while during other periods of the year, daily R_s was relatively similar (and close to zero) in the marginal areas of both disturbed and undisturbed bogs (Fig. 4 in **IV**). Median peat respiration rates for the vegetation period were between 0.5 and 1 g C m^{-2} day^{-1} in plots L2, K2 and L2 and slightly above 2 g C m^{-2} day^{-1} in plots S1.2 and S1.2 (Fig. 5a in **IV**). Since soda lime method may underestimate high CO_2 fluxes (Ewel et al. 1987; Janssens et al. 2000), our estimates of R_s are rather conservative, and C losses from drained sites could be even larger. Nevertheless, R_s values from the pristine Laeva site were comparable to those reported by Salm et al. (2012), who measured two central Estonian bogs using the closed chamber method in the same year as our study. The emissions from Selisoo bog attained somewhat higher values compared to the drained sites by Salm et al. (2012).

Contrary to our expectations, peat respiration rates at the site with the most severe anthropogenic impact on vegetation (Kalina) were not significantly

higher than at Laeva (Table 3 and Fig. 4 in **IV**). However, the variability of R_s rates increased as a result of disturbances, considered both within and between plots. At Kalina the difference between plots K1 and K2 in median R_s during the growing season was more than twofold, although the plots were located less than 100 m apart. Apparently the high variability of R_s in disturbed sites was related to the varying WTD, as R_s was strongly dependent on the hydrological conditions in the plot (Figs. 6a and 7a in **IV**). Consistent with our finding, Salm et al. (2012) detected an effect of disturbance (drainage) on vegetation composition, but not on CO_2 and CH_4 emissions, since WTD was equivalent at disturbed and pristine sites.

Both sites in Selisoo bog, Seli1 and Seli2, exhibited significantly higher R_s values than at Laeva, despite the fact that Seli2 was also undrained (Table 3 in **IV**). Approximately 10 cm difference in average WTD between Seli2 and Laeva sites was caused by a natural stream in the former area, where amelioration of surrounding areas might also have increased runoff (Niinemets et al. 2011). The difference in WTD did not cause changes in species composition or productivity in plot S2.2, but median R_s was more than 2-fold higher than at the pristine site, which may induce substantial differences in C balance. Our data are insufficient to estimate the net ecosystem productivity of the plots, as we neither measured heterotrophic and autotrophic respirations separately nor attempted to estimate belowground productivity (except for large tree roots). Therefore, we cannot say whether the increase in R_s derives from decomposition of ”old” peat or increased root respiration and decomposition of fresh litter. A new study has been initiated to investigate further the changes in peat accumulation rate at Selisoo during the previous century (Karu et al. 2014).

Methane emissions from our measurement sites ranged from –5.9 to 126.7 mg C m^{-2} day^{-1} (Fig. 5b in **IV**). These values are at the lower end of the range of emission rates recorded in pristine and drained northern forested bogs (Bubier et al. 1995; Nykänen et al. 1998; Blodau 2002; Minkkinen and Laine 2006; Salm et al. 2012). Since our measurements did not cover a full growing season and were relatively infrequent, we may have missed periods with highest fluxes. The highest emission rates were recorded at the undisturbed Laeva and Seli2 sites, especially in the L2 plot, which contained a high biomass of *Eriophorum vaginatum* L. (Table 4 in **IV**), a species known to enhance methane efflux (Frenzel and Karofeld 2000; Nykänen et al. 2002; Minkkinen and Laine 2006; Salm et al. 2012). Methane emissions from Seli1 were significantly lower, while Kalina was positioned between the two groups (Kruskal-Wallis rank sum test on log-transformed data, multiple comparisons of mean ranks; $p < 0.05$), because emissions in plot K2 were similar to the levels observed at Laeva. Plot mean CH_4 emission rates were nonlinearly and negatively correlated with WTD (Fig. 6b in **IV**), but this relationship was significant only at soil temperatures above 10°C (Spearman $r_s = –0.66$; $p < 0.05$). Also, laboratory studies suggest that methane production is highly temperature-sensitive and is low within a range of 0–10°C (Dunfield et al. 1993). Even in pristine bogs, methane emissions comprised only a minor fraction of the total C

emissions and therefore did not substantially affect the C balance of the ecosystem. On the other hand, the global warming potential calculated on a 100-year basis is 25 times higher for CH_4 compared to CO_2 (IPCC 2007). Therefore methane emissions can still contribute significantly to the total greenhouse gas budget in plots L2 and K2.

We found that peat C emissions in our poorly drained sites were strongly related to WTD, while other anthropogenic factors, like air pollution, were less important. It is difficult to disentangle the share of individual anthropogenic activities on the lowering of the water table in Kalina and Selisoo, but the impact of water pumping from underground oil shale mine on WTD in Kalina bog was probably minor, due to the geological properties of peat and underlying mineral layers (Hang et al. 2009). However, roads through Kalina bog that generate high variability in WTD and C cycle within the bog were related to oil shale mining. Our study sites in Selisoo were situated in the western part of the bog, where drainage was caused mainly by former peat mining (Seli1) and amelioration of the surrounding areas. In the eastern part of Selisoo that Estonia underground oil shale mine is approaching, the drainage effect of underground mining might be probable in the future (Hang et al. 2009). Permanent plots and wells have been set up in Selisoo to monitor the effect of underground mining on vegetation and WTD (Hang et al 2012). According to our results, species composition of the bog vegetation is not a good indicator of C fluxes in case of moderate drainage, as it is not sufficiently sensitive to changes in hydrological conditions. However, both average WTD and stand structural parameters (BA, *V*) in wooded bogs could potentially serve as indicators of changes in peat R_s rates (Fig. 7 in **IV**), although underlying relationships require further investigation based on larger datasets.

4. CONCLUSIONS

The following conclusions can be drawn and further research needs identified based on this thesis:

1. Forest plantations show remarkably good growth on calcareous and stony substrate of oil shale mining spoil. Scots pine plantations, which dominate the afforested areas, have the potential to accumulate over 130 t C ha^{-1} less than 40 years after afforestation. Most of the sequestered carbon is accumulated in tree stems, but soils also contribute a significant part (up to ~25%) to the total stand carbon stock. Because of the high variability of the parent substrate, there are still large uncertainties concerning the average rate of SOC accumulation in reclaimed oil shale opencasts. In addition, the impact of tree species on the rate of carbon accumulation on oil shale mining spoil and the partitioning of sequestered carbon among different ecosystem compartments need further research.
2. Among 30-year-old plantations established on mine spoil, alder and silver birch stands are the most productive, although differences between the plantations in growing stock and soil depth were statistically not significant due to the high variability within site types. Broadleaved trees also enhance soil development on both mine spoil and semicoke dumps more than conifers. Spontaneously developed stands, which are dominated by birches and willows, have similar soil depth and comparable tree growth rates as Scots pine plantations. Therefore, spontaneous succession may be considered a viable alternative to reclamation in exhausted oil shale mines.
3. Planted tree species affect the development of understory on mine spoil, primarily through changes induced in soil nutrient status, which vary according to tree species, and create recruitment limitations for the herbaceous vegetation. Broadleaved trees, especially alders, tend to enhance the growth of herbaceous species, whereas conifers suppress it. Different herbaceous species are favored by certain tree species leading to significant differences in species composition of the herb layer between the site types. The herb layer formed in alder stands is most distinct from the others, but highest number of species can establish in spontaneously developing stands. As a practical implication for restoration, the vegetation and landscape in a reclaimed opencast could be diversified by planting various tree species and letting natural succession occur in small areas within plantations.
4. On semicoke hills, stand age and planted tree species have no significant effect on species composition of the ground vegetation. Herb layer is dominated by ruderals species and generalists, although seven orchid species were also found. Successional changes in vegetation take probably more time on semicoke hills compared to the oil shale opencasts due to more adverse growing conditions.
5. Alkaline air pollution emitted by oil shale fuelled power plants has induced the establishment of forest species in the marginal areas of bogs. In addi-

tion, both increased influx of nutrients through air pollution and lowering of the water table contribute to increased growth of the tree layer and decreased productivity of mosses, which may lead to a shift in the functioning of bog ecosystems in the long term.

6. Contrary to our expectations, there are no significant differences in peat respiration rates and methane fluxes between heavily disturbed (Kalina) and pristine (Laeva) bog sites due to the high within-site variability of emissions in the former. In a moderately disturbed site (Selisoo), peat respiration rates are significantly higher compared to the pristine bog. Whether this indicates also increased peat mineralization rates needs further research. Peat carbon gas emissions in poorly-drained bog margins are sensitive to fluctuations in WTD; the impact of air pollution on peat respiration rates and methane emissions is less important. Average WTD and stand structural parameters (BA, *V)* can serve as relevant indicators of carbon fluxes from peat for similar ecosystems.
7. Forest functioning and ecosystem services, especially in terms of productivity and carbon sequestration, recover relatively fast in exhausted oil shale opencasts. By contrast, bog ecosystems have not recovered from past anthropogenic disturbances such as high levels of alkaline air pollution and are faced with new threats. If underground oil shale mining brings about changes in their hydrological conditions, it may have a serious impact on the carbon cycle of bogs and initiate their succession toward forest-like ecosystems. Changes in the functioning of bog ecosystems due to past and future disturbances in northeastern Estonia need further attention and interdisciplinary research efforts.

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SUMMARY IN ESTONIAN

Ökosüsteemide areng inimtegevusest mõjutatud Kirde-Eesti tööstuspiirkonnas: tehismaastike metsad ja rabad

Inimühiskonna majanduslik areng ja heaolu sõltuvad suurel määral looduslikest ressurssidest. Inimkonna kasvu ja tehnoloogilise arengu tõttu on jätkuvalt kasvamas maavarade kaevandamise ja tööstuse surve keskkonnale, ent samas on hakatud rohkem tähelepanu pöörama ka nende keskkonnamõjude leevendamisele. Üks olulisemaid kaevandamisega kaasnevaid keskkonnamõjusid on looduslike- ja kultuurmaastike asendumine uute tehislike pinnavormidega, nagu karjääripuistangud, tööstusjäätmete kuhilad ning allmaakaevanduste poolt põhjustatud maapinna langatused. Rikutud alade taastamine, arvestades peale majanduslike ka ökoloogiliste põhimõtetega, suurendab meie loodusliku kapitali, s.h. teenuseid, mida taastatud ökosüsteemid inimkonnale pakuvad.

Kaevandamiseelse ökosüsteemi taastamine ammendatud karjäärides on haruldane, kuna reeglina on seal keskkonnatingimused oluliselt muutunud, eeldused taimede kasvuks halvad ning sellega kaasneksid suured kulud. Üks levinumaid võtteid ammendunud kaevandusalade taastamiseks on puistangute tasandamine ning metsastamine. Istanduste rajamine soodustab metsakoosluste arengut ning võib läbi puiduproduktsiooni mõjutada positiivselt kogu regiooni majandust (Singh et al. 2002). Samas on üheliigiliste istanduste rajamist ka kritiseeritud, kuna neist kujunevad metsakooslused kipuvad olema madala stabiilsuse, ressursside kasutamise efektiivsuse ning bioloogilise mitmekesisusega (Lugo 1997). Seetõttu soovitatakse kaevandamisega rikutud alade taastamisel kasutada rohkem looduslikke protsesse ning neisse võimalikult vähe sekkuda (Tischew and Kirmer 2007; Prach and Hobbs 2008; Hobbs and Cramer 2008; Walker et al. 2014).

Eestis on põlevkivi kaevandatud umbes 1%-l maismaa pindalast (~430 km^2). Kirde-Eesti põlevkivimaardla põhjaosas – kuni 30 m sügavuseni – kaevandatakse põlevkivi karjäärides ning lõuna pool, kus tööonduslikud lasundid paiknevad sügavamal, allmaakaevandustes. Kui hetkel on karjääridest ja allmaakaevandustest saadava toodangu suhe umbes 50:50, siis tulevikus on oodata allmaakaevanduste osatähtsuse tõusu, kuna kergemini kättesaadavad varud hakkavad juba ammenduma. Põlevkivi kasutatakse kõige enam elektrienergia tootmiseks tahkekütusel töötavates elektrijaamades, aga samuti toodetakse sellest õli, tsementi ja erinevaid keemiatooteid. Põlevkivi baasil töötavad elektrijaamad on suurimad õhusaasteallikad Eestis, kuigi olukord on alates läinud sajandi 90ndatest aastatest oluliselt paranenud (Link 2014). Samuti pärineb põlevkivitööstusest 80% kogu Eesti jäätmetest ning suurem osa ohtlikest jäätmetest (Hermet 2014). Vanad poolkoksihoidlad Kohtla-Järve ja Kiviõli õlitehaste lähikonnas olid tõsisteks reostusallikateks, mistõttu on nad

praeguseks suletud. Paraku hävis sulgemistööde käigus ka suurem osa neile istutatud taimestikust.

Karjääriviisilise põlevkivi kaevandamisega on praeguseks rikutud üle 140 km^2 maad. Karjääripuistangute metsastamine algas 1959. aastal, kõige laiemalt on selleks kasutatud harilikku mändi (~85% metsastatud aladest). Teised enam levinud puuliigid on arukask, harilik kuusk, erinevad lehiseliigid ning sanglepp (Kaar 2010). Umbes 200 ha puistangutest jäeti looduslikule uuenemisele, mis võimaldab võrrelda looduslike protsesside käigus arenevaid ökosüsteeme inimese poolt metsastatud aladega. Puistangute rekultiveerimine on seni lähtunud pigem majanduslikest huvidest – suurendada puiduproduktsiooni, kui ökoloogilise taastamise põhimõtetest, seetõttu on rekultiveeritud karjäärides vähe uuritud selliseid aspekte nagu süsiniku salvestamine uute metsakoosluste poolt ning bioloogilise mitmekesisuse taastumine.

Kõige tõsisemaks allmaakaevandamisega kaasnevaks mõjuteguriks taimekooslustele on hüdroloogilise režiimi muutused. Eriti võivad kuivenduse tõttu kannatada veetaseme muutuste suhtes tundlikud ökosüsteemid, näiteks sood, mis katavad umbes 40% Kirde-Eesti pindalast. Veetaseme langus võib käivitada protsessid, mis suunavad soode arengu metsalaadseteks ökosüsteemideks (Cedro and Lamentowicz 2008), ning soodustada turba lagunemist, millega kaasneb suurenev CO_2 emissioon turba lasundist (Moore and Dalva 1993; Martikainen et al. 1995; Silvola et al. 1996). Samas metaani (CH_4) emissioonid kuivendamise tagajärjel vähenevad (Martikainen et al. 1995; Nykänen et al. 1998; Minkkinen et al. 2007; Salm et al. 2009). Erinevalt madal- ja siirdesoodest on rabad ainult sademeveelise toitumisega, mistõttu on nad väga tundlikud õhusaaste suhtes. Põlevkivielektrijaamadest pärinev aluseline õhusaaste ja toitainete sissekanne on Kirde-Eesti rabades põhjustanud turbasammalde produktsiooni langust ning kiirendanud rabamändide kasvu (Karofeld 1994, 1996; Pensa et al. 2004a; Kaasik et al. 2008). Vaatamata elektrijaamadele paigaldatud efektiivsele tolmufiltritele ja õhusaaste olulisele kahanemisele viimastel aastakümnetel, ei ole seal rabadele omane taimestiku liigiline koosseis veel täielikult taastunud (Karofeld et al. 2007; Paal et al. 2010).

Käesoleva töö esmaseks eesmärgiks oli kirjeldada taimkatte ja mulla arengut ammendatud põlevkivikarjääris ja poolkoksimägedel ning hinnata, kuidas neid protsesse mõjutab rekultiveerimiseks kasutatud puuliigi valik. Lisaks võrreldi liigilist mitmekesisust ja produktiivsust looduslikult taastuvatel ning metsastatud aladel. Teiseks eesmärgiks oli analüüsida muutusi, mida on rabaökosüsteemides põhjustanud aluseline õhusaaste ning veetaseme alanemine.

Töös kontrolliti järgmisi hüpoteese:

1. Metsastamine kiirendab taimkatte produktiivsuse taastumist ammendatud põlevkivikarjäärides. Suurem osa uute metsade poolt salvestatud süsinikust akumuleerub puutüvedesse, ent ka mulla roll on oluline.
2. 30-aastastes lepikutes ja kaasikutes on puiduproduktsioon kõrgem ning mulla viljakus ja -sügavus suuremad kui sama vanades männikutes, mis

karjäärialadel domineerivad. Looduslikule uuenemisele jäetud alad taastuvad oluliselt aeglasemalt kui istandused.

3. Metsastamiseks kasutatud puuliigid mõjutavad alustaimestiku arengut nii karjääripuistangutel kui poolkoksimägedel.
4. Inimtegevus – aluseline õhusaaste, metsakuivendus, turbakaevandamine ning põlevkivi allmaakaevandamine – põhjustab muutusi Kirde-Eesti tööstuspiirkonnas asuvate (puis)rabade taimestikus ja süsinikuringes ning tõukab nende arengut metsakoosluste suunas. CO_2 vood turbast on häiringutega rabas suuremad ja CH_4 emissioonid väiksemad kui ilma otsese inimmõjuta aladel.

Karjääripuistangute arengut uuriti aastatel 2002–2005 Narva karjääris, kus mõõdeti taimkatte ja mulla parameetreid kokku 80 proovialal (10 igast kasvukohatüübist): (*i*) hiljuti (~5 a. tagasi) tasandatud puudeta puistang, (*ii*) 10-, (*iii*) 20- ja (*iv*) 30-aastased männikud, (*v*) 30-aastased lehise (siberi ja euroopa lehis), (*vi*) -arukase ja (*vii*) -lepa puistud (hall- ja sanglepp) ning (*viii*) looduslikult uuenenud alad. Kolmes männikus uuriti detailsemalt süsiniku akumuleerumist. Kohtla-Järve poolkoksimägedes uuriti kooslusi 76-l proovialal aastatel 2004 ja 2006.

Inimtegevuse mõju rabadele vaadeldi Kalina rabas, mis paikneb allmaakaevandatud alal, on mõjutatud freesturbakaevandamisest ning aluselisest õhusaastest, ning samuti Kirde-Eestis paiknevas Selisoos, mis asub elektrijaamadest kaugemal. Selisoos paiknes kaks uurimisala: Seli1 raba põhjaosas asub looduslikult taastunud endisel pätsturba kaevandamise alal ning Seli2 raba läänеosas on ilma vahetu inimmõjuta. Looduslikus seisundis võrdlusala (Laeva) paikneb Alam-Pedja looduskaitsealal. Kõigil uurimisaladel hinnati taimestiku liigilist koosseisu ja aastast produktsiooni ning kord kuus mõõdeti veetaset, mullahingamist ja metaani emissiooni. Välitööd viidi läbi 2009. aasta jooksul.

Kuna karjääripuistangutel domineerivad hariliku männi puistud, esindab doktoritöös uuritud männikute aegrida tüüpilist ökosüsteemide suktsessioonilist trajektoori ammendatud põlevkivikarjäärides. Uuritud puurinde ja mulla parameetritest muutuvad vanusega kõige enam puidutagavara ja mulla paksus – mõlemad näitajad on omavahel ka tugevalt korreleeritud. Vaatamata toitainetevaesele ja kivisele puistangute substraadile osutuvad Narva karjääri rajatud istandused väga produktiivseks. Samas ei ole puistute kasvukiirus ja tihedus puistangutel ühtlane, mistõttu 30-aastate männikute puidutagavara varieerus vahemikus 100 kuni 360 m^3 ha^{-1}. Muld hakkab puistangutel kujunema teisel aastakümnel peale metsastamist. Orgaanilise aine akumulatsiooni ja puistu arenguga kaasnevad ka muutused mulla keemilises koostises: mulla lämmastikusisaldus kasvab vanusega oluliselt, samas kui pH, kaaliumi ja fosfori sisaldus kahanevad.

Kolm hariliku männi puistut, mille süsinikuvaru uuriti detailsemalt, olid puidutagavara ja mulla parameetrite poolest lähedased kõigi mõõdetud männikute keskmisele. Süsinikuvaru kasvu peetakse üheks olulisemaks ökosüsteemi taastumise indikaatoriks kaevandamisega rikutud aladel (Frouz et al. 2009).

Käesolevas töös leiti, et põlevkivikarjääridesse rajatud männikud võivad salvestada üle 130 t ha^{-1} süsinikku vähem kui 40 aasta jooksul peale nende rajamist. Nagu eeldati, akumuleerub enamus süsinikust puude tüvedesse. Mulla osatähtsus kogu metsa süsinikuvarust kasvas 5%-lt noorimas, 15 aasta vanuses puistus, kuni 23%-ni 36 aasta vanuses puistus. Kuna karjääripuistangutel on mullakeskkond ruumiliselt väga heterogeenne, vajavad täpsemad mulla süsinikusalvestamise hinnangud laialdasemaid uuringuid, samuti oleks vaja täiendavalt uurida erinevate puuliikide mõju süsiniku akumuleerumise kiirusele ning selle jaotumisele ökosüsteemi erinevate komponentide vahel.

30 aastat peale metsastamist annavad kõige enam puiduproduktsiooni arukase- ja sanglepapuistud, samas istandustevahelised erinevused puidutoogas ja mulla paksuses ei olnud metsatüübisisese suure varieeruvuse tõttu statistiliselt olulised. Sellegipoolest võib tulemuste põhjal järeldada, et lehtpuupuistud, eriti kõrgeima mulla toitainesisaldusega lepikud, stimuleerivad mulla ja alustaimestiku arengut karjääripuistangutel ja poolkoksimägedel enam kui männikud. Looduslikult uuenenud metsades, kus puurindes domineerivad arukask ja remmelgad, on mulla paksus ja puiduproduktsioon võrreldavad männikutega. Seega võib väita, et looduslik uuenemine on põlevkivikarjääride puhul arvestatav alternatiiv metsastamisele.

Peale puiduproduktsiooni ja süsiniku salvestamise uuriti kaevandamise tõttu tekkinud tehismaastikel ka taimestiku arengut ja selle sõltuvust istutatud puuliigist. Leiti, et rohurinde liigiline koosseis männikute aegreas on eri vanusejärku puistutes oluliselt erinev. Noorimates arengujärkudes domineerivad jäneskastik (*Calamagrostis epigeios*) ja paiseleht (*Tussilago farfara*), vanusega nende osatähtsus väheneb ning valgusnõudlikud pioneerliigid asenduvad järkjärgult varjutaluvate liikidega. Vanadele metsadele omaste liikide (Hermy et al. 1999) osatähtsus 30-aastastes männikutes oli 18%. Lisaks puistu vanusele mõjutab rohurinde liigilist koosseisu ka puuliik – istutatud puud muudavad keskkonnatingimusi enda ümber, mõjutades sellega alustaimestiku liikide kasvamamineku võimalusi. Kuna erinevate puuliikide keskkondakujundav toime on erinev, järgneb iga puuliigi istutamisele talle iseloomuliku alustaimestiku kujunemine. Doktoritöös leiti, et teistest kõige enam erineb lepikute rohurinde liigiline koosseis, samuti on seal suurim rohurinde produktiivsus. Kõige rohkem liike võib leida looduslikult uuenenud puistutes, kus puurinde ja mikroreljeefi suure mitmekesisuse tõttu leidub sobivaid elupaiku suuremale arvule liikidele kui inimese poolt rajatud istandustes. Tööst saab teha praktilise järelduse: liigilise mitmekesisuse suurendamiseks karjääripuistangutel tuleks metsastamisel kasutada senisest enam lehtpuid ning jätta osa alasid looduslikule uuenemisele. Poolkoksimägedel puistu vanus ega puuliik alustaimestiku liigilisele koosseisule olulist mõju ei avalda ning rohurindes domineerivad ruderaalid ja ilma kindla elupaigaeelistuseta generalistid. Ilmselt kulgeb suktsessioon poolkoksil aeglasemalt kui karjääripuistangutel, kuna tingimused on seal taimede kasvuks keerulisemad.

Põlevkivielektrijaamadest pärinev aluseline õhusaaste on toonud kaasa olulisi muutusi Selisoo, eriti aga Kalina raba taimestiku liigilises koosseisus –

rabale iseloomulikud liigid moodustavad Kalina proovialade taimestikust vähem kui 60%. Nii õhusaaste kui veetaseme alanemine on põhjustanud Kalina ja Seli1 uurimisaladel puurinde produktsiooni kasvu ja samblarinde produktsiooni vähenemist, mis pikemas perspektiivis tähendab nende muutumist funktsionaalselt teistsugusteks ökosüsteemideks. Samas, vaatamata meie eeldustele, ei erine CO_2 ja CH_4 emissioonid turbast kõige suurema inimmõju all olnud Kalina uurimisalal oluliselt looduslikus seisundis Laeva alast. Selisoos on CO_2 emissioonid turbast siiski oluliselt kõrgemad kui Laevas – samuti vahetu inimmõjuta alal. Kas suurenenud CO_2 voogudega kaasneb ka intensiivistunud turba lagunemine, vajab veel edasist uurimist. Süsinikuemissioonid turbast nõrgalt kuivendatud uurimisaladel sõltuvad tugevasti veetasemest, samas kui õhusaaste mõju pole ilmselt oluline. Doktoritöö tulemused näitavad, et puisrabades ja suhteliselt kõrge veetasemega rabametsades võiks turbast pärinevate süsinikuvoogude hindamiseks kasutada keskmist veetaset ja/või puistu rinnaspindala, kuid täpsemate kvantitatiivsete seoste kindlakstegemiseks on kindlasti vaja suuremat andmehulka Eesti erinevatest soodest.

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PUBLICATIONS

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Publications:

Karu, H., Pensa, M., Rõõm, E.-I., Portsmuth, A., Triisberg, T. 2014. Carbon fluxes in forested bog margins along a human impact gradient: could vegetation structure be used as an indicator of peat carbon emissions? *Wetlands Ecology and Management* 22: 399–417.

Pisek, J., Lang, M., Nilson, T., Korhonen, L., **Karu, H.** 2011. Comparison of methods for measuring gap size distribution and canopy nonrandomness at Järvselja RAMI (RAdiation transfer Model Intercomparison) test sites. *Agricultural and Forest Meteorology* 151: 365–377.

Pensa, M., **Karu, H**., Luud, A., Kund, K. 2010. Within-species correlations in leaf traits of three boreal plant species along a latitudinal gradient. *Plant Ecology* 208: 155–166.

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Pensa, M, Luud, A., **Karu, H.**, Vaht, R. 2005. Taimkatte taastumine põlevkivikarjäärides: istutatud ja looduslikult uuenenud puistute võrdlus. [Reclamation of vegetation in oil shale opencasts: A comparison of plantations and naturally developed stands]. In: Sammul, M., Lõhmus, A. (Eds.). Ökoloogiline taastamine [Ecological Restoration]. *Yearbook of the Estonian Naturalists' Society* 83: 96–112 (in Estonian with English summary).

Karu, H., Luud, A., Pensa, M., Rull, E., Vaht, R. 2005. Taimkatte arengust põlevkivikarjääride taastamisel [Development of vegetation on reclaimed oil shale opencast mines]. In: Liblik, V., Punning, J.-M. (Eds.). *Keskkond ja põlevkivi kaevandaminde Kirde-Eestis*. [Environment and oil shale mining in North-East Estonia]. Tallinn University, Institute of Ecology, Tallinn, pp. 121–133 (in Estonian with English summary).

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Karu, H., Pensa, M., Alliksaar, T., Avel, E. Anthropogenic impact on vegetation and peat accumulation in two Estonian bogs. Poster. 8th International Symposium on Ecosystem Behaviour Biogeomon 2014, Bayreuth, Germany, 13–17 July 2014.

Karu, H., Pensa, M., Rõõm, E.-I., Portsmuth, A. Does vegetation structure in disturbed bogs indicate alterations in bog functioning? Poster. 56th Symposium of the International Association for Vegetation Science (IAVS 2013), Tartu, Estonia, 26–30 June 2013.

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Karu, H., Pensa, M., Rõõm, E.-I. Carbon emissions from bog margins along a human impact gradient. Suuline ettekanne. Centre of Excellence in Environmental Adaption – ENVIRON Workshop "Carbon, nitrogen and water fluxes in ecosystems: main emphasis on FAHM-experiment", Tartu, Estonia. 29 November 2011.

Pensa, M., **Karu, H.** Carbon emissions from bogs along a human-impact gradient in Estonia. Poster. COST Action FP 0601 Forest Management and the Water Cycle, 7th workshop. Helsinki, Finland, 6–8 September 2010.

Pensa, M, **Karu, H**. Reforestation in Estonian oil-shale opencasts: a challenge for restoration ecology. Poster. COST Action FP 0601 Forest Management and the Water Cycle, 5th workshop. Sede Boqer, Iisrael, 10–12 November 2009.

Pensa, M., **Karu, H.**, Luud, A., Rull, E., Vaht, R. The effect of planted tree species on the diversity of herbaceous vegetation in a reclaimed oil-shale opencast in Estonia. Oral presentation. Adapting Forest Management to Maintain the Environmental Services. Koli, Finland, 21–24 September 2009.

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Karu, H., Pensa, M., Rõõm, E.-I., Portsmuth, A. Does vegetation structure in disturbed bogs indicate alterations in bog functioning? Stendiettekanne. 56th Symposium of the International Association for Vegetation Science (IAVS 2013), Tartu, Eesti, 26.–30. juuni 2013.

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Pensa, M., **Karu, H.** Carbon emissions from bogs along a human-impact gradient in Estonia. Stendiettekanne. COST Action FP 0601 Forest Management and the Water Cycle, 7th workshop. Helsingi, Soome, 6.–8. sept. 2010.

Pensa, M, **Karu, H**. Reforestation in Estonian oil-shale opencasts: a challenge for restoration ecology. Stendiettekanne. COST Action FP 0601 Forest Management and the Water Cycle, 5th workshop. Sede Boqer, Iisrael, 10.–12. nov. 2009.

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- ETF6677 (M. Pensa). Kasvukoha viljakuse ja klimaatiliste tingimuste mõju boreaalse metsaökosüsteemi igihaljaste taimeliikide lehetunnuste muutlikkusele
- SF0282119s02 (V. Liblik). Tehismaastike looduslik-antropogeensed arengusuunad Kirde-Eesti põlevkivikaevandamise piirkonnas
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2012 Kursus "Summer School on Radiocarbon in Ecology and Earth System Science". Max-Planck Institute for Biogeochemistry, Jena, Saksamaa.

2006 Doktorantide kursus "Tree canopy – structure and functioning – from below and above". Põhjamaade võrgustik "Carbon Dynamics in Managed

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2006 Kursus "Concepts within Ecosystem Modelling and their Application". Joensuu Ülikool, Joensuu, Soome.

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– Ühingu "Society for Ecological Restoration" liige

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